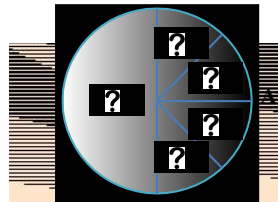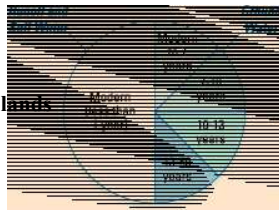**Piedmont & Uplands**

Atlantic Coastal Plain

Incorporating Lag-Times Into The Chesapeake Bay Program

STAC Workshop Report
October 16-17, 2012
Annapolis, Maryland

STAC Publication 13-004

Table of Contents

Executive Summary

A STAC-sponsored workshop entitled,―Lag-times in the Watershed and Their Influence on Chesapeake Bay Restoration‖ was held on October 16-17, 2012 at the Sheraton Hotel in Annapolis, Maryland. The workshop was attended by 48 invited participants, including 9 presenters. The workshop defined―Lag-times‖ as the time elapsed between installation or adoption of point or nonpoint source (NPS) control practices and the effect of that practice to the targeted water body.

The goals of the workshop were to bring together a diverse set of experts who would suggest ways in which the concept of lag-times could best be represented in simulation models of the Bay and its watershed, to suggest directions for data collection and research to improve the understanding of lag-times, and how to communicate implications of lag-times in the Bay with the public and stakeholder groups.

Lag-time Overview

The general concept of lag-times is that the impact of events currently taking place in the watershed, including changes in land use and management practices, changes in point source loading rates, and specific natural events such as extreme floods or droughts, will not be entirely reflected in changes to water quality for periods of years to many decades. Conditions in the Bay at any given time are a result not only of the current human activities in the watershed (point source inputs, NPS inputs, and the current management strategies that reduce these inputs), but also of a legacy of past activities in the watershed, some of them extending back decades or even centuries in time. Thus, the present degraded conditions in the Bay and its tributaries are a result of a long history of activities on the landscape, and management actions taken today will have positive impacts at time scales ranging from days to centuries. The potentially long periods of these lag-times do not constitute an excuse for inaction, but they do constitute a reason for being patiently realistic about the time-scale for observing results.

Water quality monitoring and Best Management Practice (BMP) effectiveness. Consideration of lag-times also raises issues related to water quality monitoring, and evaluation of BMP effectiveness. If understanding of the linkages between BMPs and water quality (in the tributaries and in the Bay) is to improve, it is critical to develop and maintain much better data about the timing and nature of the BMPs and other factors that change pollution inputs (this applies to past changes as well as to on-going and future changes). The scientists who study the behavior of the system will also need to use models that explicitly evaluate observed water quality as a function of hydrologic conditions (including major flood and drought events) and the time history of the inputs of pollutants to the system. The only way that can be done is through models that explicitly consider lag-times in sediment and groundwater movement and explicitly consider the storage and release of pollutants from the watershed (from floodplains, soils, reservoir sediments, and groundwater).

Cause of lag-time for nutrients and sediment. Lag-times are the result of various natural fate and transport processes that occur across the landscape and within streams between the pollutant sources and the receiving water bodies. The following is a list of processes important to the three water quality pollutants that are simulated in the Chesapeake Bay Watershed Model (CBWM) and the focus of restoration efforts:

1.   Nitrogen (N) moves through the watershed in many forms and through many pathways from its varied sources (atmospheric deposition, fertilizer, manure, or point source discharges) to receiving waters. The main transport of nitrogen in the watershed occurs underground, as dissolved nitrate is moved through the soil vadose zone by percolating water and into slow moving aquifers. Transport also occurs through surface runoff in dissolved, particulate, or organic forms and associated episodic cycles of in-channel deposition, scour, and re-deposition.

2.   Phosphorus (P) (from chemical fertilizer, manure, or point source discharges) can follow all of these same pathways as nitrogen, but subsurface transport in the dissolved state is much less important for phosphorous. Phosphorus transport occurs primarily during episodic storm events that produce runoff and cause sediment-bound phosphorus to be carried into streams where they can be desorbed through biogeochemical processes or deposited, only to be re-suspended and re-deposited by subsequent storm events.

3.   Sediment (derived from natural weathering as well as a multitude of human-related activities) moves in a highly episodic manner with each particle likely to be deposited and re-suspended numerous times from its initial movement from the landscape until it is eventually deposited in the Bay mainstem.

Lag-time is the time gap between BMP implementation and delivery of the full water quality effect to the Bay. Hence, lag-time will influence public perception of progress towards meeting the Chesapeake Bay Total Maximum Daily Load (TMDL) targets. Water quality monitoring data used in the calibration of the CBWM will not be totally reflective of management practices in place because of lagged reductions in delivered pollutants. From an implementation planning perspective, however, this is a less serious issue since implementation focuses on the installation and initiation of field practices and BMPs. TMDL target loads are set based on load limits from the estuarine model, and watershed implementation plans (WIPs) are based on the level of BMP implementation, not actual in-stream pollutant reduction, even though that is the eventual desired outcome. Therefore, although both the Bay TMDL and the WIPs are lag-time independent, progress in water quality improvements is highly dependent on, and sensitive to, lag-times in the system.

The statements above can be supported by considering the current modeling suite in the CBP, i.e., that lag-time is not explicitly considered in the current version 5.3.2 of the Chesapeake Bay Watershed Model (CBWM) and the Chesapeake Bay TMDL, but it could affect future

implementation strategies. Currently, any BMP incorporated into the CBWM has an immediate effect on model estimates of nutrients or sediments delivered to receiving waters, presumably leading to lower in-stream nutrient and sediment concentrations. However, since lag-times are associated with virtually all NPS BMPs, it is unrealistic to expect that actual water-quality improvements will match model results for scenarios with those BMPs in place. Even if the model representations of BMP effectiveness were highly accurate, the modeled improvements would take many years to be realized. Example illustrations from either simplified spreadsheet analyses or CBWM scenarios are needed to illustrate how the consideration of lag-times influence the improvement of water quality due to BMP implementation over time, both locally and to downstream receiving waters, such as the Chesapeake Bay.

Implications and Recommendations

BMP effectiveness. The importance of BMPs in nutrient and sediment reduction is critical to the Bay's restoration, and BMP implementation has become the tracked outcome in the CBP's 2-year milestones. Hence, identifying BMP maturation and effective operational periods for inclusion in the CBP models should be an expanded priority. Recent CBP model revisions have begun including BMP efficiencies as a function of precipitation amount and age of the BMP and this should continue, with further considerations of landscape type and geographic location (i.e., site specific issues) for use in the model.

Point-to-nonpoint nutrient trading. One of the most important implications of lag-times is for point-to-nonpoint nutrient trading programs. A basic requirement of trading programs is that the allowance held by the point source and the offset sold by the NPS must be ecologically equivalent both spatially and temporally. If lag-times are accounted for, and permits must still be met annually, the economics driving the decision of point-source permit holders about whether to participate in trading or to install advanced treatment technology will favor the advanced treatment technology. Since installing enhanced nutrient treatment facilities requires an irreversible capital investment, once installed, the point source may not enter the market in future years and the demand for offsets from agriculture may be permanently reduced. In order to adjust for the effects of lag-times, existing trading programs could be revised to incorporate forward markets to efficiently allocate reductions over time, but that would add an extra layer of complexity making point-to-nonpoint source trading even more difficult to implement. Alternatively, trades could be restricted to those BMPs known to produce relatively rapid responses in receiving waters.

Inventory of agricultural and urban BMPs. A major baseline data need for assessing the impacts of lag-time is a comprehensive local inventory of all agricultural and urban BMPs, including performance characteristics. Although this issue has already been high-lighted by many Bay Program partners, it bears repeating here, as the effects of any actions related to lag-time are impossible to quantify when the actions themselves have not been fully quantified.

As part of a BMP inventorying process, efforts to enable data sharing from federal sources related to farm-specific BMPs should be expedited, but in a manner that protects client privacy. Nutrient management should also be expanded from a mere planning tool to an accountability mechanism. Measures, previously unaccounted for, such as farm and recreational ponds and streambank restoration, need to be included as well.

As urban areas begin their BMP inventory efforts for stormwater management, standardization and dissemination of accounting methods, inventory procedures, and performance verification are needed to facilitate widespread and rapid adoption of consistent data to feed into a region-wide database.

Using supplemental multiple models with the CBWM. The current CBWM is a complex model that provides insight into the broad-scale distribution and delivery of nutrients and sediment from a wide variety of sources, but it was never designed to calculate field-scale impacts of site-specific BMPs. Supplemental models should be used to inform the CBWM on processes not currently simulated, to facilitate insights into lag-time, and to provide site-specific targeting of BMP placement for more effective load reduction. Supplemental models should be used to explore an improved representation of sediment and attached nutrient storages, rather than the current use of sediment delivery ratios to edge-of-stream and the in-stream delivery factors, because sediment storage, rather than transport, appears to be the primary regulator of downstream sediment delivery. This same idea can be applied to the dissolved forms of nitrogen and phosphorus. For example, it is the overloading of P in the surface layer of some agricultural soils (particularly those receiving manure that is not worked into the soil) that drives P loss from agricultural lands. This suggests that the direction of research in watershed modeling should take a mass balance perspective rather than just modeling transport with loss terms. Models should consider the sources, the transformations, the long-term and short-term storage, and ultimately, transport out of the system. The models also need to consider the transformation of attached particulate nutrients into dissolved forms and their subsequent behavior.

Supplemental models should also be used to explore improved representation of surface-groundwater interchange, as delivery of dissolved nutrients is largely controlled by groundwater flow. Additional topics related to lag-times that should be explored by supplemental models in order to inform the CBWM include the impacts of improved representation of lower order streams, stream channel erosion, and groundwater and reservoir dynamics. Future revisions to the CBWM should incorporate lessons-learned from the supplemental models, where possible.

Framework for within-watershed hydrologic, sediment, and nutrient interactions. A conceptual framework needs to be developed that encompasses the interactions between floodplains, stream channels, and sediment storages. Major data needs identified in the workshop centered on developing comprehensive spatial and temporal sediment budgets to better quantify the size and distribution of various sediment storages within any given watershed. Tools to help in this effort

include particle size distribution studies, isotope tracer studies, sediment coring in reservoirs, use of sediment "fingerprint" techniques to determine the role of various sediment sources, and coupled sediment-nutrient analyses. Research needed at a very local scale includes long-term monitoring at various points along a flow path to the nearest stream (including the impacts of riparian buffers); monitoring of nutrient movement and partitioning between the soil surface, the unsaturated zone, and baseflow in physiographic provinces and landscape settings; and repeat (3-year interval) geo-referenced sampling of phosphorus in the upper few centimeters of soil in fields with high soil test P, as this is the major soil P storage locale, as well as the source of event-driven P delivery. Research needed at the regional scale includes inflow-outflow monitoring of larger reservoirs to improve their representation in the CBWM and groundwater flow path delineation to better understand regional variability in pollutant delivery from these sources.

Water quality monitoring implications. A new focus is also needed in our collective monitoring strategies, consistent with the new conceptual framework that includes the impact of storages on lag-time. There is a distinct difference between results monitored in a demonstration watershed and those expected from a larger geographic area. Funding is scarce for both experimentation and for additional monitoring, but the general public and legislators demand results and accountability. Given these constraints and demands, we recommend providing guidance to new monitoring efforts to explicitly evaluate hypotheses needed to guide restoration, BMP implementation, and land planning in a holistic manner. Complementary to this strategy, an expanded dialogue is needed between the scientists who assess water quality (through monitoring and data analysis) and modelers, both to improve calibration of the models and to improve our understanding of the changes taking place in the watershed.

Importance of public and stakeholder communications. Improved communication with the public is needed to distinguish between BMPs that may be expected to show relatively immediate water quality benefits and those whose impact may not be seen for some time. In a similar manner, additional ecosystem system benefits, beyond those anticipated in the Bay itself, need to be highlighted, such as wildlife habitat and improved recreational opportunities, which should also result from implemented BMPs.

Communicating the general magnitude of lag-times for various BMPs is very important for maintaining public support for Bay restoration. We need to emphasize that lag-times are part of all natural systems and that they vary widely for different practices. For example, shorter lag-times are associated with practices that occur close to water resources such as ―livestock exclusion from streams‖ or ―upgrading nutrient removal from treatment plants,‖ while longer lag-times are associated with practices that occur farther from streams and which involve nutrients transported by slower processes (e.g., groundwater and cyclic in-stream deposition/suspension of sediment). Information about lag-times must be used to inform the adaptive management process, to educate the public about setting realistic restoration

expectations, and to assist local managers in more appropriate selection of control measures that will produce the desired short-term and long-term effects necessary for Bay restoration.

Introduction

Restoration programs for the Chesapeake Bay began in the mid-1980s. These programs have some elements that can be characterized as being successful, while other elements have seemingly made slower progress toward their goal. This irregular progress toward Bay restoration can be frustrating to the general public. However, this frustration is often due to an under-appreciation of the time it takes for natural ecosystems to respond to changes in management practices.

―One important reason nonpoint source watershed projects may fail to meet expectations for water quality improvement is lag-time. Lag-time is an inherent characteristic of the natural and altered systems under study that may be generally defined as the amount of time between an action and the response to that action‖ (Meals et al. 2010). For the purposes of this workshop, lag-time is defined as the time elapsed between installation or adoption of point or NPS practices and the observed effect of that practice in the targeted water body. There are other lags to eventual system response (Fig. 1) such as the time to identifying the problem (e.g., N and P inputs upstream control estuarine phytoplankton production downstream, D'Elia et al. 1992) and then the identification of the strategy to address the problem (e.g., detergent phosphate ban in MD, addition of biological nitrogen removal [BNR] at waste water treatment plants), which were not addressed in this workshop.

General Components of Lag-time

The main physical system components of lag-time include the time required for an installed practice to produce an effect at the source, the time required for the effect to be delivered to the water resource, and the time required for the water body to respond to the effect. This workshop primarily focused on the first two components. Additionally, although project management and the planning process contribute to the perceived lag between action and result, and monitoring limitations contribute to a lag in detecting change, they are not part of the delays between action and response in the physical system, which is the focus of this workshop. The physical system components will be briefly explored in order to better illustrate the overall challenges that lag-times add to restoration of a large natural ecosystem like the Chesapeake Bay, and to better define those components addressed by this workshop.

The time required for the adopted management practice, or the installed process, to become fully functional and to produce a desired effect at the implementation site (Component 3, Fig. 1) is the initial component of lag-time examined at the workshop. This stage has been described in detail by Meals et al. (2010) and primarily involves the activation time needed for the physical or biological processes. An example of a short lag-time for this component in the Bay would be

establishment of stream livestock exclusion fencing that produced measurable results in less than one year (Meals et al. 2001). An example of longer lag-times would be the use of a rye cover crop to lower nitrate-N concentrations in shallow groundwater in Maryland, which produced significant reductions within 2-3 years of planting, but took about 10 years to reach full effectiveness (Staver and Brinsfield 1998). Newbold et al. (2008) also reported that a newly established riparian forest in Pennsylvania took about 10 years to reduce shallow groundwater nitrate-N.

The time required for the effect to be delivered to the targeted water body (Component 4, Fig. 1) was another element discussed at the workshop. This component can be broken down into three factors (Meals et al. 2010): the transport route to a water body (e.g., direct deposition, surface runoff, groundwater recharge, etc.), the distance transported, and the rate of travel along each path (e.g., fast for artificial drainage, moderate for subsurface flow through soils, very slow for regional aquifers, etc.). It is also useful to distinguish the three general classes of substances that will be considered in this workshop, specifically: water soluble nutrients, mainly nitrate-N; water sparingly-soluble nutrients, mainly P; and suspended sediment. The usefulness of these substance classes is that they are directly related to their transport media, with soluble compounds like nitrate transported through groundwater, while sparingly-soluble compounds like P and suspended sediment are transported primarily in surface runoff followed by interactions within the stream (Ator et al. 2011).

Once P and sediment reach the stream they can be repeatedly adsorbed, deposited, and re-suspended as they move downstream toward the receiving water body. This intermittent cyclic stream transport is often associated with episodic high flow during storm events, which creates challenges for monitoring and lengthens the time of pollutant delivery to downstream water bodies. An example of the importance of storm events in transporting P and sediment is the data summarized by Hirsch (2012) from the Conowingo Dam, which shows that while the 2011 Tropical Storm Lee event produced less than 2% of the total annual stream flow during 2002-2011, it accounted for 22% of the P and 39% of the suspended sediment transported past the Conowingo stream gauge during the same time period. Soluble compounds, like nitrate, are primarily transported through groundwater at recharge rates similar to groundwater flow. Estimates of ground water ages and their contributing distribution to stream baseflow in the upper Eastern Shore are: less than 7 years old – 36%, 7-13 years old – 10%, 13-50 years old – 25%, and greater than 50 years old – 29% (Sanford 2012); corresponding estimates for the Chesapeake Bay above the fall line by Phillips and Lindsey (2003) are: less than 7 years old – 25%, 7-13 years old – 50% and 13-50 years old – 25%. In either case, it is apparent that groundwater flow can result in substantial delayed delivery times for nitrate from any on-land practices that modify surface inputs to the groundwater flow system.

The last component of physical system lag-time (Component 5, Fig. 1) is the time it takes for the water body to respond to the new conditions. Although this component was not a major part of

this workshop, it is important to realize that it is a significant factor and that it is also quite variable. This is the component, however, that needs most attention for educating the public on the time it will take for detecting the water quality improvements that citizens and managers expect to see. This component depends on such factors as those above, as well as flushing rates
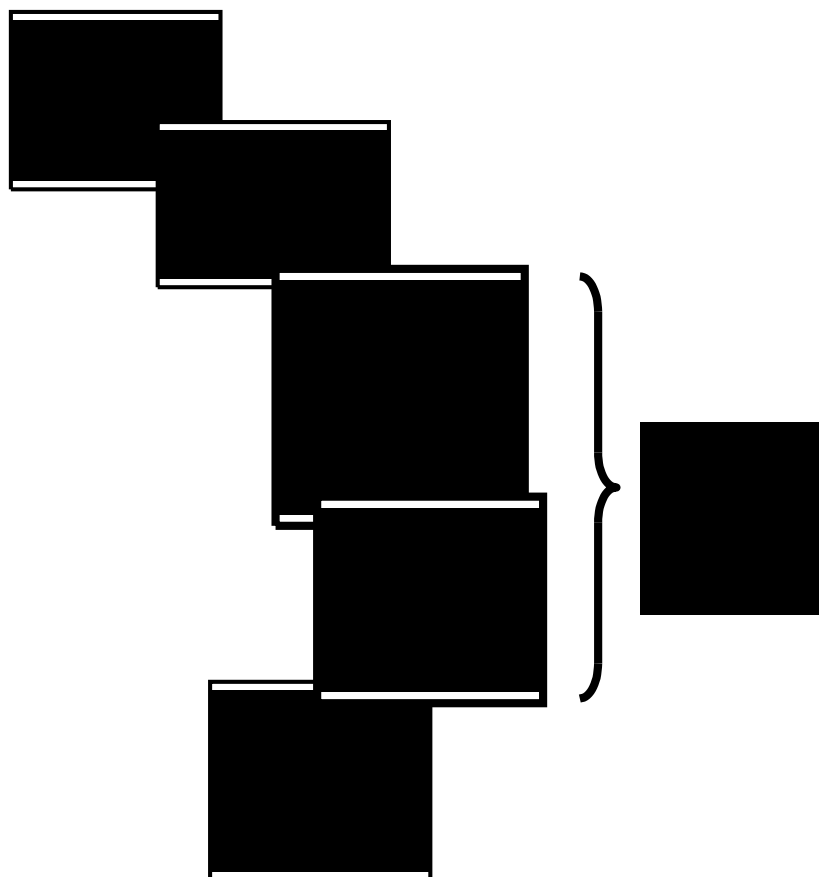
Figure 1. Summary of major lag-time components experienced within the Chesapeake Bay restoration program. Overlapping areas in components indicate mutual areas of interest or activity. Lag-times within components are commonly observed ranges, rather than absolute minimum to maximum values.

In the receiving waters, the indicator evaluated, the impairment involved (particularly where biological responses are desired), and monitoring frequency and duration. For example, reducing bacterial contamination may be accomplished in less than a year (Meals et al. 2010),

while changes targeted to restoring habitat like submerged aquatic vegetation or three dimensional oyster reefs may involve several years as shown by Batiuk et al. (1992) and North et al. (2010), respectively.

Overview of the Workshop

The STAC-sponsored, two-day workshop entitled, ―Lag-Times in the Watershed and their Influence on Chesapeake Bay Restoration‖, was held on October 16-17, 2012 at the Sheraton Hotel in Annapolis, Maryland. A copy of the full workshop agenda is included in Appendix A. The goal of the workshop was to bring together a diverse set of experts who could suggest ways in which the concept of lag-times could be represented in simulation models of the Chesapeake Bay watershed. The approximately 40 workshop attendees were encouraged to prepare for the workshop by reading a report from a previous STAC-sponsored workshop on lag-times (Korcak et al. 2005) and two reviews of research related to lag-times in water quality response to best management practice implementation (Meals et al. 2010, NNPSMP 2008). A full list of workshop participants can be found in Appendix B.

Presentations were given on various modeling possibilities for incorporating the effects of lag-times, various current fate and transport research related to lag-times, and implications for lag-times on water quality trading policy. Two-page summaries of each of the nine presentations are included in Appendix C. Three breakout groups were organized around the topics of:

  • Group A: Processes associated with erosion, storage, and re-entrainment of sediment and associated nutrients;
  • Group B: Processes associated with transport, reaction, and storage of nutrients in their dissolved form; and
  • Group C: Considerations of lag-time in the context of regulation, enforcement, pollutant trading, and public perception.

These three breakout groups initially met late on the first day and then again during most of the second day discussing the following set of key questions:

  1. What new data collection, data analysis, and research are needed at the scale of individual management actions (a single BMP implementation)?

  2. What new data collection, data analysis, and research are needed at the scale of river reaches, reservoirs, floodplains, wetlands, and aquifers?

  3. What new approaches to modeling should be developed and/or enhanced to better understand and predict lag-times?

4. Are there modifications (perhaps post processing) of existing watershed models that could adjust their results to better accommodate lag-times? Would implementing these likely be worthwhile?

5. Are there some broad general statements that STAC can make to the Bay community about the typical lag-times for sediment, nitrogen, and phosphorus associated with broad categories of BMPs that would be applicable over the entire Bay watershed (or significant portions of the Bay watershed)? Is it even useful to try to do this?

6. Does the consideration of lag-times matter to the implementation of policies such as load allocations or effluent trading? How should these policies deal with the issue of lag-times?

7. What, if anything, can or should be done about improving the understanding by the public and public officials regarding lag-times?

This report summarizes the discussions from the workshop centered around each of these seven questions and concludes with a set of recommendations and implications regarding data collection, research, model development, policy development, and public communications related to the representation and communication of lag-times in Chesapeake Bay restoration efforts.

Steering Committee Members

The workshop steering committee was comprised of the following STAC members and Chesapeake Research Consortium staff:

Bob Hirsch, Chair

Russ Brinsfield	David Sample
Matt Ellis	Kevin Sellner
Natalie Gardner	Don Weller
Kurt Gottschalk	Claire Welty
Jack Meisinger	Gene Yagow
Marc Ribaudo	Weixing Zhu

1. What new data collection, data analysis, and research are needed at the scale of individual management actions (a single BMP implementation)?

New Data Collection and Analysis

More Comprehensive BMP Inventories

One major source of new data collection needed at a very local scale, already highlighted by various partners in the Bay Program, is that of a comprehensive inventory of all agricultural and urban BMPs. Knowing where they are and how they are performing is a pre-requisite to estimating lag-times for these control measures. While USGS and USDA have an agreement through their response to the President's Executive Order that farm-specific BMPs will be made accessible in some aggregated fashion to protect client privacy, site-specific data in a restricted access mode are still preferred, where possible. Details of the inventory should include: location (as specific as possible); extent; position in the landscape; age; installation; performance; and maintenance characteristics. Crops planted, seeding and harvest dates, and yields as well as amounts and dates of fertilizer/manure applications would allow assessment of the actual effects of nutrient management, not just of what was planned.

These data should be expanded to include the existing BMPs currently absent from state cost-share tracking efforts, including farm ponds, stream restoration, and voluntary BMPs of any nature. Nutrient management (NM) plan details are currently also considered proprietary, but their details are essential to understanding the impacts of this widespread planning tool. Furthermore, follow-through is needed to monitor actual impacts of NM, so that the observed weather conditions and interim farmer decisions on actual amounts of fertilizer applied and yields harvested (referenced above) can be used to assess pollutant reductions.

The process of inventorying urban BMPs has begun in conjunction with various localities' attempts to document progress already made towards their TMDL Watershed Improvement Plan goals, and with other localities in documenting their compliance with Municipal Separate Storm Sewer System (MS4) permit requirements. Standardization and dissemination of accounting methods, inventorying procedures, and performance verification are needed to facilitate widespread and rapid adoption of consistent data to feed into a region-wide database.

More Resolved Spatial Representation

Additional ‗local' data needs include improved spatial representation of potential nutrient and sediment sources and storages. Better information about near-stream geomorphology would be helpful in tracing the paths of more recent pollutants and their connection with the stream through shallow interflow. For example, high resolution topography (Light Detection and Ranging or LIDAR, see below) could be utilized to characterize near-stream conditions, to identify areas with larger slopes, and to better delineate and characterize floodplains. In addition

to their effects on transport pathways and time scales, these features also affect the distribution of natural attenuation processes (e.g., sediment trapping or nitrate reduction) that can moderate responses to up-gradient BMPs.

Additional data needs identified in the workshop include:

- Comprehensive spatial and temporal sediment budget analyses to evaluate volumetric distributions of potential sediment storages and sinks.

- Particle size distribution data (and associated P) in fields, channels, stream banks, stream deposition sites, wetlands, and floodplains, as well as in the water column, both pre- and post-BMP implementation. Particle size, stream velocity, and channel turbulence determine the relative rates of particle deposition, and whether the fine particles remain suspended for delivery to the bay or if they become transient or fixed in long-term deposits (for example, millennial storage in floodplains (Pizzuto 2012)).

- Isotopic tracer studies to calculate residence and travel time distributions, and identify sediment source and sink areas.

- Reservoir core data, including soil texture and composition, to better understand historical legacy influences. Cores from farm ponds and mill ponds could provide additional information.

- Coupled sediment and nutrient analyses to determine delivery mechanisms from various storages and sources, in order to answer questions such as—are legacy stream banks contributing high P?‖

New Research

New research needs at a very local scale identified in the workshop include:

- Long-term monitoring of single BMPs at various points along a flow path from source to the nearest stream to ascertain where reductions and additions of different pollutants take place and how long after implementation it takes to maximize pollutant reduction (BMP maturation).

- Monitoring of nutrient movement and partitioning throughout the hydrologic cycle, i.e., from the atmosphere to the soil surface and biota through the unsaturated zone to the water table and groundwater to stream baseflow in various landscape settings. This should be done in a number of physiographic provinces and landscape settings, including riparian buffers. These results are important for answering such questions as—are riparian buffers important for denitrification?‖ and—do buffers largely affect nitrate from older legacy groundwater sources or from younger surface runoff from the fields?‖ Could the Sanford et al. (2012) model be used to further investigate nitrate sources mediated by buffers? Hydrochemical flowpath studies should investigate the importance of buffers as a carbon source for potentially enhancing in-stream denitrification.

- Repeated, geo-referenced soil phosphorus (P) analyses in fields with high soil test P that are being managed for P reduction (discussions at the STAC Small Watershed Monitoring Design Workshop recommended repeat sampling at 3-year intervals over a decade or more (Weller et al. 2010)) to document lag-times involved in reducing soil P storages. P in the upper centimeter is critical as well as this is the source of event-driven P delivery and sub-surface (root zone and below) P stores.

- Evaluation of multiple environmental tracers (chemicals, isotopes) to determine flow paths, travel times, and reactivation of contaminants across multiple time scales including hourly, event-scale, seasonal, inter-annual, and decadal.

In order to provide consistency and to maximize monitoring resources, guidance should be developed to provide local watershed managers with advice on where and when monitoring should be used to attempt to detect progress, and which parameters and monitoring strategies are most important for monitoring various BMP installations, landscapes, and pollutants. Some of this information is in recent reports, Mostaghimi et al. (2007), Williams et al. (2010), Sellner et al. (2011), and Wicks et al. (2011).

2. What new data collection, data analysis, and research is needed at the scale of river reaches, reservoirs, floodplains, wetlands, and aquifers?

The most important consideration at the scale of river reaches, reservoirs, floodplains, wetlands, and aquifers are the storages of pollutants in transit to receiving waters. Understanding reservoir sediment dynamics is essential because of the transition from nutrient and sediment sinks in their early lives, to nutrient and sediment sources as they reach their maximum storage capacity. For example, filling of reservoirs may complicate the interpretation of BMP progress in the watershed, as they will initially increase lag-time, and then precipitously produce loads that have no relationship to the current state of upland BMPs (see Hirsch 2012).

An analogy was made during discussions at the workshop comparing pollutant loads and load reduction to the current United States economic downturn and recovery. In the same way in which these loads were not introduced over one‒term,‖ the reductions will also take longer than one‒term‖ to be achieved.

While new data and research are needed, an underlying theoretical framework is also required, so that in the future, any stream channel reach and its associated watershed of varying stream order can be assessed, control variables put in place, and reasonable landscape scale predictions made.

As alluded to in several general model recommendations in Band et al. (2008), a conceptual framework for sediment transport is needed that encompasses the interaction between floodplains, stream channels, and stream restoration and how sediment storages change before and after stream restoration. The size of the floodplain relative to the exchange flux is one of the explanatory causes of lag-time that can be readily monitored from remote sensing. The current

availability of LIDAR in many locations (Pennsylvania, Baltimore metro area, Eastern shore, coastal zones of VA) could be used for many purposes to observe change and characteristics on the landscape. Another application of the conceptual framework is to understand sediment behavior at nested scales, so that managers can make better informed decisions about which sources to control and which BMPs might be most effective. Hopefully, such a framework could also be used to better inform our understanding of legacy sediments and attached pollutants and how different particle sizes are affected by fate and transport mechanisms. The conceptual framework, in turn, could then provide a more rational basis for organizing the monitoring program at multiple scales in a nested fashion to make it easier to trace sediment movement from upstream to tidal outlets.

It is also important to incorporate legacy pollutants in modeling efforts. Knowledge of the amounts of nitrogen and phosphorus in storage in various parts of the landscape is crucial to projecting the trajectory of water quality conditions under any scenario of future land management changes.

New Data

Comprehensive Reservoir Inventory

A detailed, spatially-explicit inventory of reservoirs in the Chesapeake Bay watershed would provide better information to evaluate related transport and storage characteristics. Because most small reservoirs are not monitored at the inlets or outlets, upstream and downstream monitoring are needed to assess their sediment trapping capabilities, as well as to quantify dynamics when their available volume for storage of sediment and attached pollutants is approached. Alternatively, bathymetry analysis of the reservoir could be combined with soil core data to document the evolution of a reservoir's storage history. Similar improved spatial representation is also needed for rivers and streams in terms of cross-sectional areas.

Additionally, major land use changes can also greatly influence shifts in flow regimes and flow paths and should be considered together with other dynamic factors when trying to assemble a conceptual model of the various influences on lag-times in a specific watershed and the different effects of urban vs. forest vs. agricultural lands. Current land use-land cover data provide limited information detailing land management activities and identifying potential nutrient/sediment sources. In urban areas, stream channel incision and erosion cause most of the sediment loading to local waterways, which is not currently represented in the CBWM, and therefore makes it difficult to link control measures with load reductions. Stream incision also can promote transmission of agricultural solutes such as nitrate beneath riparian soils at short time scales and with minimal reaction (e.g., Böhlke et al. 2007, K. Belt in Groffmann 2012).

New Research

Spatial-Temporal Variation in Flows

One important component of understanding lag-times is to recognize the diversity of flow paths that deliver water to streams under different flow conditions (e.g., low base flow, high base flow, storm flow, etc.) and how they change in relative importance at various time scales. Information on both surface and groundwater flow paths are needed to fully understand linkages among various storage sinks across the landscape and how they influence residence times.

General Regional Characteristics

Research needs to be coordinated to see if it is possible to quantify relative lag-times associated with broader physiographic regions, and then drill down to more site-specific influences. One approach is to evaluate loads and trends across the watershed for spatial patterns. A complementary approach, currently in progress at the USGS, is to characterize processes at selected representative sites and then determine how to regionalize (interpolate, extrapolate) on the basis of climate, geology, and land use. Without fully understanding groundwater lag-times and their interaction with surface water, estimates of sources and appropriate control strategies might be misguided. Alternatively, the control strategies might be appropriate, but predicted outcomes may be misleading, leading to inappropriate expectations. Since a framework is needed that encompasses long periods of time, corresponding data collection efforts need to be longer in duration in order to fully capture lag-time effects.

An important first step would be to compile existing information about flow paths and residence times and determine if some classification can be made on the basis of regional hydrogeology (topography, geology). Some relevant research in different hydrogeomorphic regions of the Chesapeake Bay watershed is summarized in Lindsey et al. (2003) for groundwater flow paths and in Focazio et al. (1998) for springs. Surficial unconsolidated coastal-plain aquifers such as parts of the Delmarva Peninsula may discharge mixtures of groundwater in which the age distribution decreases relatively smoothly in abundance from young to old, as illustrated in Figure 2. Watersheds with shallow soil and regolith over crystalline bedrock such as the Mahantango watershed in Pennsylvania can have more complex age distributions with large fractions of very young water coupled with substantial fractions of older mixed ages, as illustrated in Figures 3 and 4. Other examples include stratified coastal-plain aquifers (e.g., SERC, Böhlke et al. 2007), where stratification of hydraulic conductivity, groundwater age, and chemistry result in seasonally varying proportions of younger and older discharge; karst terrains, such as the Great Valley, where discharge can include rapid conduit flow and slower matrix flow; and other crystalline rock settings, etc. (Lindsey et al. 2003). New data and analysis are needed to test whether results from local studies can be regionalized for watershed-scale modeling. Improved models are needed to incorporate variations in hydrogeology and to include all parts of the hydrologic cycle at a range of time scales.
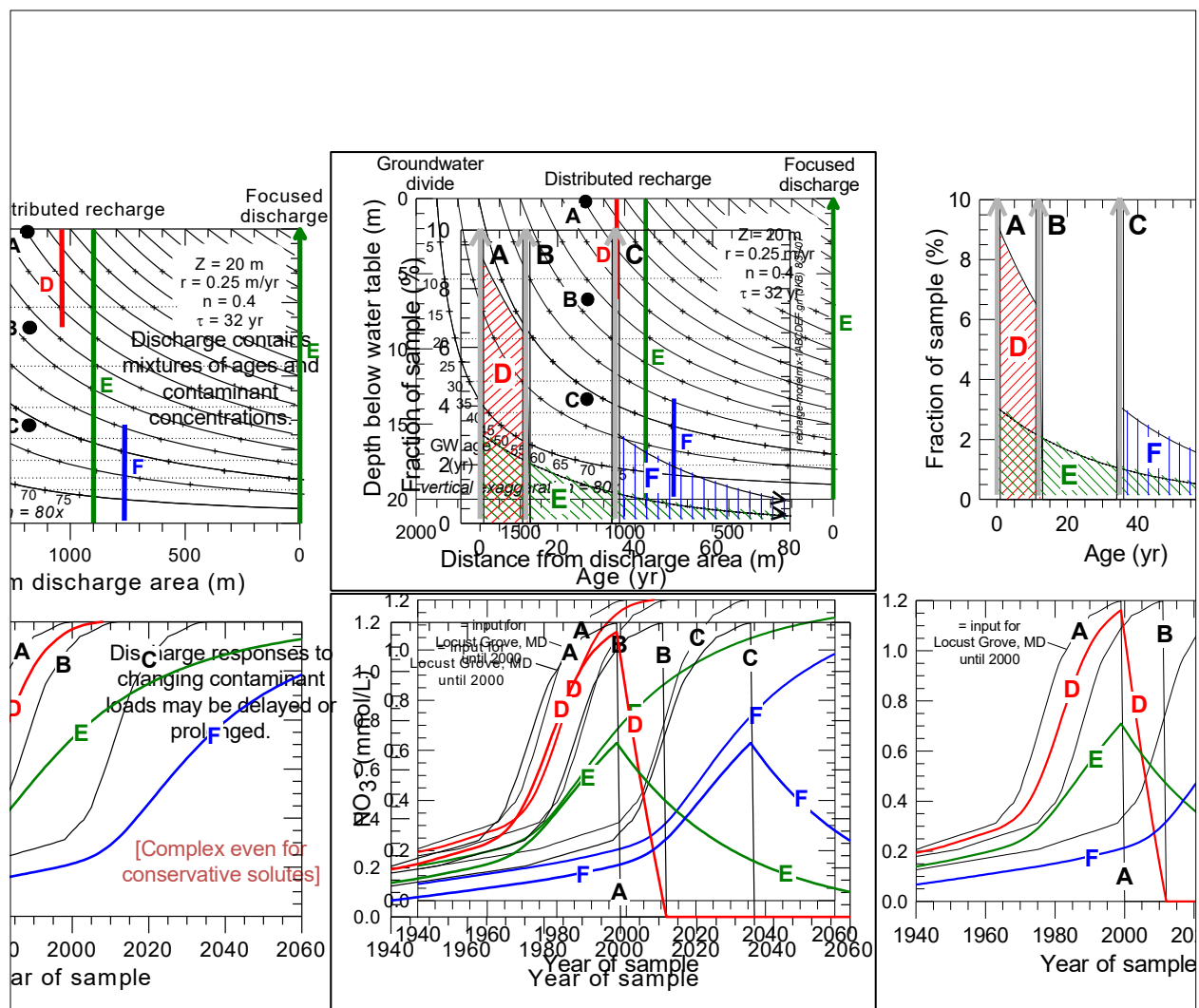
Figure 2. Delmarva groundwater characteristics (coastal plain example), based on Locust Grove, MD (Bohlke and Denver 1995), illustrating effects of different age distributions on contaminant responses in a surficial aquifer with uniform properties and distributed recharge (exponential model). The upper panel shows groundwater flow lines, age contours, and hypothetical configurations of well screens in the aquifer. The bottom panel shows the modeled time history of nitrate concentration at each well, given the input history at the water table shown by the curve labeled A (derived from groundwater data at Locust Grove). Wells sampling single points in the aquifer (such as A, B, or C) are most likely to have a limited range of groundwater age in their discharge, so contaminant response mimics the change at the water table with a fixed delay time (lag-time). In contrast, wells with long screens (such as D, E, or F) sample groundwater with wider ranges of ages, so responses are more gradual and complex. The response curve for well E could also represent discharge from the aquifer to a stream, represented by the upward arrow at the far right edge of the upper panel (figures modified from Bohlke 2002).

Apply convolution integral to particle bins
Rapid partial response (young fractions)
Slow complete response (old fractions)
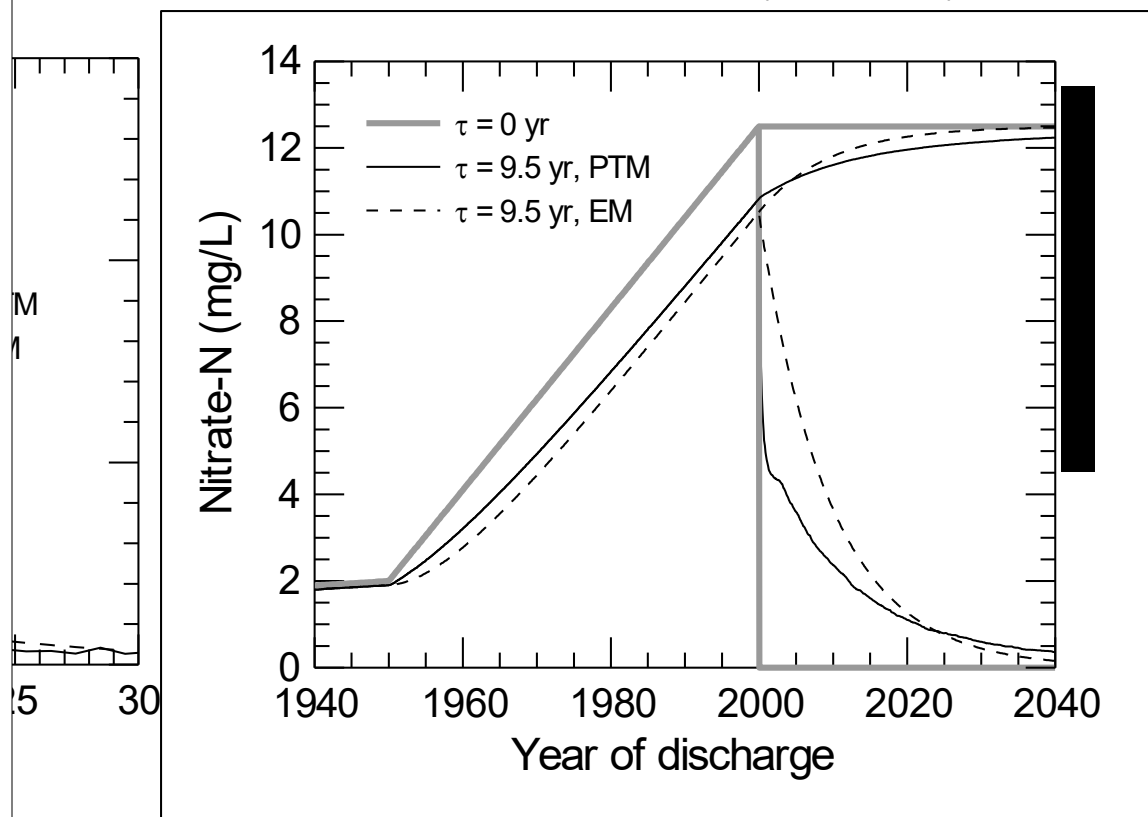
Figure 3. Illustration of "lag-time" effects based on a field study and particle-tracking groundwater flow model (PTM) for the Mahantango Creek watershed, PA (weathered fractured crystalline rock example) (Lindsey et al. 2003). It illustrates a system with contrasting hydraulic properties in the shallow and deep parts of the groundwater flow system. Gray lines represent two hypothetical input histories. Increasing nitrate concentration in recharge water from 1950 to 2000, then two scenarios from 2000 forward. Upper line is a constant input from 2000 forward. Lower line is a total shutoff of nitrate input in 2000. Solid black line represents PTM simulation of discharge nitrate response to the two scenarios. Rapid response of the shallow parts of the flow system show about half of the total response takes place within the first 1 to 2 years, while the slow response from the deep part accounts for the long tail. Dashed lines show corresponding model results for a hypothetical aquifer with an exponential age distribution (EM) with the same overall mean age as in the PTM example (τ is the mean age of the water). The exponential response curves are different from the layered system response curves, despite having the same mean age (figure modified from Lindsey et al. 2003).

Mahantango Creek watershed, Pennsylvania
Fractured rock aquifer
Hydraulic conductivity decreases downward

Age distribution in groundwater discharge
(particle tracking model)
Large young component with long tail
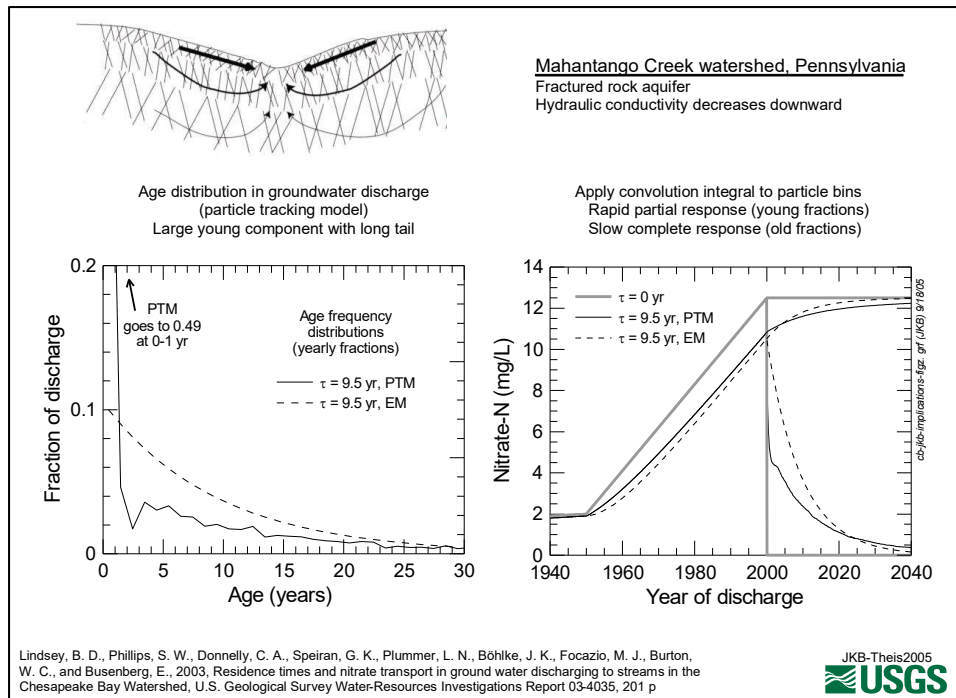
Apply convolution integral to particle bins
Rapid partial response (young fractions)
Slow complete response (old fractions)

PTM goes to 0.49 at 0-1 yr

Age frequency distributions (yearly fractions)
— τ = 9.5 yr, PTM
- - - τ = 9.5 yr, EM

Fraction of discharge

Age (years)

τ = 0 yr
τ = 9.5 yr, PTM
τ = 9.5 yr, EM

Nitrate-N (mg/L)

Year of discharge

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Lindsey, B. D., Phillips, S. W., Donnelly, C. A., Speiran, G. K., Plummer, L. N., Böhlke, J. K., Focazio, M. J., Burton, W. C., and Busenberg, E., 2003, Residence times and nitrate transport in ground water discharging to streams in the Chesapeake Bay Watershed, U.S. Geological Survey Water-Resources Investigations Report 03-4035, 201 p

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Figure 4. Weathered fractured crystalline rock example-based on Mahantango, PA (Lindsey et al., 2003). This illustrates a system with contrasting hydraulic properties in shallow and deep parts of the groundwater flow system. Solid lines in plots show discharge age distribution on the left and discharge response on the right to increasing N followed by constant or decreasing N at the water table for this layered system. For hypothetical rise and then shutoff of N input at the water table, rapid response from the shallow part accounts for about half the recovery, while slow response from the deep part accounts for the long tail. Dashed lines show corresponding model results for a hypothetical aquifer with an exponential age distribution with the same overall mean age. The exponential response curves are different from the layered system response curves, despite having the same mean age (figures modified from Lindsey et al. 2003).

Current estimates from two different methods by Sanford (2011) suggest 60-70% contribution of groundwater to baseflow. Additional groundwater modeling studies that will help shed light on these basin estimates are being conducted by USGS, moving from the Delmarva to the upper Potomac basin with a focus on shallow and deep groundwater flow paths and delivery to streams. This new phase of regional distributed flow modeling could highlight some of the contrasts between flow systems in previously modeled unconsolidated sediment coastal plain aquifers (Delmarva) and karst and weathered crystalline rock settings (upper Potomac). To capture the

range of important time scales of change in groundwater delivery and stream loads, more continuous sensors should be deployed in streams and springs for various chemical parameters such as specific conductance, turbidity, dissolved oxygen, nitrate, etc. (e.g., Pellerin et al. 2012).

The spatial arrangement of BMPs on the landscape definitely influences response time for the implemented practice. Until watershed studies can be performed to help us better understand treatment trains, high resolution models could be used in small watersheds to identify important landscape features that increase or decrease pollutant loading and thus be used to inform larger, less resolute models such as the CBWM (see Band et al. 2008). Another interaction between modeling and monitoring could be to derive more effective sampling designs to monitor denitrification. For instance, the Penn State Integrated Hydrologic Model (PIHM; Duffy et al. 2012) could be used to perform numerical experiments and identify timing and locations where denitrification would be more likely to occur, increasing the effectiveness of scarce monitoring resources.

Multiple Site Examinations

Currently, research is often designed around individual watershed or study goals without consideration of the ‗bigger‗ picture or of how the research results feed into a larger context, resulting in many studies that are not comparable to others; lag-times vary from one small watershed to the next. Further, it is not economically feasible to examine each individual watershed as implementation and accompanying monitoring cannot be everywhere at once, and there are intricacies in each individual stream. The idea of undertaking a collective demonstration project, combining both scientific and managerial efforts, could be informative, but these demonstrations could vary from place to place. Yet, there are few (if any) watershed demonstration projects that begin with examining the source(s) of sediment, targeting management actions to reduce sediment in important sources areas, and monitoring the effectiveness of these management actions on reducing sediment. State-of-the-art modeling approaches and theory should be coupled with data collection so that both inform the other. A related point was made during workshop discussions that the disagreement in research results between different research entities may be related to each entity‗s continual study of one set of watersheds or of watersheds only in one physiographic region, so that research results are interpreted in a limited context different from others, and therefore, prevent a common unveiling of overarching principles. A suggestion was made that perhaps federal agency researchers could be moved periodically to different settings within the Chesapeake Bay watershed, or alternatively, that academic researchers with a suite of skills could collaborate across basins, as well as nationally, so that researchers are exposed to different areas than the ones they have traditionally studied, allowing them to test accepted principles in different settings. Examples of such comparative studies of groundwater nitrate transport related to agriculture in contrasting hydrogeologic settings include: Lindsey et al. (2003), McMahon et al. (2008), Green et al. (2008), Liao et al. (2012), Eberts et al. (2012). In addition, many other individual USGS studies

have also been conducted with this concept embedded in project design, such as Bohlke and Denver (1995) and Bohlke et al. (2007) in the Chesapeake Bay watershed, and across the US. However, most of these studies were not aimed specifically at evaluation of best management practices, as emphasized recently in the USGS Water Science Strategic Plan (Evenson et al. 2012), and many did not include all parts of the hydrologic cycle.

Monitoring as a Tool to Evaluate Models

A new focus is needed in our collective monitoring strategies. There is a distinct difference between results monitored in a demonstration watershed and those expected from a larger geographic area. Funding is scarce for both experimentation and for additional monitoring, but the general public and legislators demand results and accountability. Given our limited resources and the urgent need to improve water resource management (e.g., the CBP set a target of restoring over 900 miles of stream buffers per year, between now and 2025; http://www.chesapeakebay.net/issues/issue/forest_buffers), it is important to consider shifting the design of monitoring programs from traditional experimental frameworks to research programs designed explicitly to evaluate the tools/models (hypotheses) used for guiding restoration, BMP implementation, and land planning (i.e., model validation). Note that there are modeling as well as monitoring tools outside of the current CBWM (e.g., Band et al. 2008), that can be used to improve understanding of underlying fate and transport processes, and then to inform the bay-wide modeling framework.

Lag-times and Historical Load Assimilation

While lag-times confound the relationship between control strategies and load reductions, longer lag-times should work in our favor by giving the riverine and estuarine systems a longer time to assimilate historical loads, while reducing our current footprint on future loading. Systems with longer residence times also tend to have damped (diluted) peak concentrations compared to input pulses on shorter time scales. Is there utility in looking at strategies that increase lag-times in order to slow down pollutant loading to the bay, through increased storage and retention time? For reactive contaminants like nitrogen, this can also promote biologic cycling and loss.

Additional Factors in Pollutant Delivery via Groundwater

It is not sufficient to approximate the mean age or average distribution of residence times of groundwater for major basins. While pollutant transport through groundwater and the response or delay (lag-time) associated with it are typically defined as a function of age distribution, Eberts et al. (2012) consider additional factors in interpreting responses of public supply wells to changing inputs of NPS contaminants to aquifers, as follows: (1) delay time before initial response, (2) recovery time (or flushing time) to reach background or some arbitrary value, and (3) dilution factor for peak discharge concentration relative to peak input concentration.

Mean age of discharging groundwater does not indicate how a receptor (well, spring, stream) will respond to a change in contaminant loading in recharge. This depends on the age distribution (also known as age frequency distribution, probability density function, travel time distribution, residence time distribution, age―histogram,‖ etc.), which describes the relative amounts of all ages in a discharge sample. This is important because discharges generally are mixtures, and mixtures with identical mean ages can transmit very different responses for the same landscape change, depending on the relative contributions of younger and older components and how variable they are.

Initial response can be immediate or delayed depending on the relative amounts of young groundwater components in discharge. If the groundwater flow system is thin, shallow, and highly transmissive, then the initial response of discharge to a landscape may be immediate (few years or less). Even in the classic―exponential model‖ for groundwater discharge from thicker unconsolidated aquifers (similar to some coastal plain models), where the mean age of discharge may be decades, young water is a substantial component of the mixture, so a partial initial response may not be delayed, although it may be more difficult to detect.

Recovery (flushing) time can be short or long depending on the age range and age distribution, and this can be independent of whether the initial response is immediate or delayed. If all discharging groundwater is young, then the time to full recovery will be short. In the exponential model, even though the initial response may be immediate, full recovery will be gradual and may take many decades (or more). A common model for watersheds underlain by shallow bedrock (e.g., Mahantango, PA example, Fig. 4) has a substantial component of young discharge through soil and regolith and another component with much longer mean age that flows through deeper low-permeability units before discharging. In this case, the initial response can be fast and relatively strong, whereas full recovery can take a long time.

Because age distributions in mixtures affect the approach to steady state, they can control the maximum contaminant concentration (or dilution factor). An aquifer with a short response time experiencing a long period of stable contaminant input will reach a steady state in which the concentration of the contaminant will be equal in recharge and discharge. In contrast, an aquifer with long response time experiencing a short period of contaminant input will not reach steady state, and the peak concentration in discharge will never be as high as the highest concentration in recharge.

The interactions of these factors (aquifer configuration, age distribution, and contaminant history) are illustrated for several representative aquifer types by Eberts et al. (2012) in Figure 5. An interactive Excel Workbook program for exploring some of these effects is described by Jurgens et al. (2012).

Real responses also will be affected by the spatial distribution of contamination on the landscape and by the distribution of denitrification or other processes along the flow paths.

Setting	Conceptual Model	Age Distributions / Cumulative Age Distributions	Breakthrough Curves for Nonpoint Source Across Recharge Area Lasting 25 Years (no degradation)	Water Quality Response to Nonpoint Source Across Recharge Area
unconsolidated (alluvial fan) sediments; long well screen (example from Modesto, California)	young water / old water (not to scale)	Particles, EP, DM	Input, Particles, EP, DM	delayed response / moderate dilution / long flush time
carbonate rocks (karst); overlain by discontinuous confining unit and unconsolidated sediments; open-hole completion (example from Tampa, Florida)	young water / very young water / old water (not to scale)	Particles, BEP, DM	Input, Particles, BEP, DM	rapid response / moderate dilution / moderate flush time following a rapid partial flush
unconsolidated (valley-fill) sediments; short well screen (example from Woodbury, Connecticut)	young water / old water (not to scale)	Particles, L	Input, Particles, L	rapid response / little dilution / short flush time
layered, confined unconsolidated sediments; confining unit short-circuited by upgradient wells completed in multiple aquifer layers (example from York, Nebraska)	young water / old water / young water (not to scale)	Particles, DM	Input, Particles, DM	delayed response / large dilution / long flush time

Eberts, S.M., Böhlke, J.K., Kauffman, L.J., and Jurgens, B.C., 2012, Comparison of particle-tracking and lumped-parameter age-distribution models for evaluating vulnerability of production wells to contamination. Hydrogeology Journal, v. 20, p. 263-282.

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Figure 5. Contrasting hydrogeologic settings with different public supply well configurations (from Eberts et al. 2012). This illustrates various age distributions and contaminant responses. Responses are characterized by: (1) delay time (lag-time) before a response occurs; (2) relative dilution of the response peak concentration in comparison to the input contaminant peak concentration, caused by mixing in systems that do not reach steady state; and (3) flushing time (recovery time), which can be long or short depending on age distribution, recovery criteria, etc.

It may be important to specify the time scale of interest. Much of what has been discussed above is related to responses that may take years to decades, such that short-term monitoring might not be sufficient to document the change. On the other hand, it may be important to understand how responses might appear at shorter time scales (hourly to seasonal) in relation to streamflow and

other factors because they could affect the types of monitoring and modeling needed to detect and quantify changes.

Sediments

Agricultural and urban land use practices are important to the delivery of sediment (and its attached nutrients) to the streams, but deposition of these sediments in stream beds and floodplains as well as subsequent remobilization of these materials in subsequent high flow events are all critical to understanding the impact of BMPs on downstream fluxes of sediment and attached nutrients. While much research has focused on the field loss of sediments and nutrients, more investigation is needed into deposition and erosion processes in the channel and riparian zone downstream. Freshwater flocculation studies that include assessment of particle size distributions and reactivity may also improve our understanding of sediment transport and storage dynamics.

An analogy appears to be in place for the region: the current sediment research community is similar to that of the nitrate community 20 years ago. New tools have been developed in recent years for "fingerprinting" sediment particles so that it becomes possible to track the material in its episodic movement downstream. Just as focused studies of nitrogen movement in watersheds has advanced greatly in the last three decades, now is the time for an increase in watershed scale study of sediment budgets coupled with models that consider the mass balance of these materials as they move through the watershed.

3. What new approaches to modeling should be developed and/or enhanced to better understand and predict lag-times?

Embedding Real-world Lags into Models

As with all models, refinements to lag-times in delivery of water and accompanying dissolved and particulate materials in the Chesapeake Bay Watershed Model (CBWM) is a goal that should be pursued to make the CBWM even more representative of the _real world.' Because the CBWM is the basis for determining nutrient and sediment responses to land use change in the region, adjusting deliveries of water and its constituents to lotic and lentic systems that best represent natural flow paths and times will provide most realistic projections of water quality to be expected from the regional implementation of BMPs, thereby providing implementers with informed expectations for identifying the effectiveness of the practices they have supported.

An expanding interest in regional modeling is quantifying/representing storage of sediment and nutrients on the land, subsurface, and receiving waters. In the current model, sediment storages are represented as gross sediment delivery ratios to edge-of-stream and then as stream delivery factors to the tidal outlets. A better representation of sediment storage throughout the system would lead to more representative downstream delivered loads (Pizzuto 2012).

For sediments, Skalak (2009) and Skalak and Pizzuto (2010) in Pizzuto (2012) argue that sediment storage is the primary regulator of sediment delivery to the bay, not transport. They have documented substantial storage of fine grain sediment in the South River, a 4th order stream of the Shenandoah River, through deposition behind fallen woody debris and in channel margins; 17%-43% of the annual suspended sediment load is stored with a mean residence time of 1.4 years. These are the deposits that 'erode' during high flow events and deliver sediments to tidal waters. Deposition rates of these fine grain sediments could be determined based on the sediment chemical composition (radionuclides Cs137 and Pb210, surface-active contaminants like mercury [Pizzuto 2012], rare earth elements [Kreider et al. 2012]), combined with results of simple hydraulic models (e.g., Narinesingh 2009 in Pizzuto 2012) to evaluate event-driven sediment dynamics. Other sediment sources, such as stream banks, can be eroded with erosion rates estimated through several techniques, such as present vs. historical shoreline photo, LIDAR image analyses, or root exposure/growth ring analysis. Transport of average-size sediments from these sources contribute to in-stream pools with little entering the tidal reaches; presumably, fine grain sediments from stream banks move downstream as noted above, cascading down-gradient behind debris and along shore margins. Partitioning transport and loads to tidal waters should be modeled as a function of particle size (Pizzuto 2012), as the finest, most reactive particles move most rapidly through storm-driven or high flow event mobilization.

A modeling component also is needed to account for changes in nutrient losses resulting from changes in nutrient application methods. The widespread adoption of no-till planting methods in the last several decades means that applied nutrients are no longer mixed into the soil as they were prior to the development of no-till practices. Nutrient placement has long been recognized as a key element of nutrient management and can rapidly change the potential for nutrient losses in storm flow. The availability of soluble nutrient forms on the soil surface or in uppermost soil layers (0-2 cm), either in chemical fertilizers or organic wastes, is a key short-term determinant of nutrient concentrations in storm flow (e.g., Staver and Brinsfield 2001, Pote et al. 2006, Verbree et al. 2010). Improved management of applied P was identified as critical in previous reviews of agricultural P management in the Chesapeake Bay watershed (Sharpley 2000), but the impacts of nutrient placement practices on both N and P loads have yet to be tracked or captured in modeling efforts. Changes in tillage methods also rapidly affect storm flow nutrient losses. But unlike nutrient placement practices, tillage methods are tracked and the effect on nutrient loads is captured generally in current modeling efforts by moving land from hi-till to low-till categories. While tillage and nutrient placement rapidly change the potential for edge-of-field nutrient losses, how rapidly these changes alter nutrient loads delivered to the Bay will vary across the Bay watershed due to the complex processes of deposition and erosion of sediments and nutrient processing in stream systems.

Yet another component of nutrient losses from cropland that needs to be covered by modeling efforts is the impact of soil P concentrations. Although not tracked systematically, soil P levels

above optimum levels needed for crop production are known to occur in areas of concentrated animal production as a result of long-term manure applications at rates equal to or greater than what is needed to meet crop N needs (Swink et al. 2009, Vadas and Sims 1998). Elevated soil P levels increase the potential for P loss by increasing the P concentration of eroded soil and by increasing dissolved P concentrations in surface runoff (Staver and Brinsfield 2001), leachate (Maguire and Sims 2002), and shallow subsurface storm flow (Kleinman et al. 2007). In contrast to the short lag-times associated with changes in tillage and nutrient application methods, much longer lag-times are associated with the edge-of-field P loss component driven by soil P concentrations. Elevated concentrations are especially expected where soil P reserves have built up to levels greater than what is needed to support target crop yields for many years or even decades of crop harvest (McCollum 1991, Kratochvil et al. 2006). In these settings, which are not spatially quantified but known to exist, short term changes in P application rates will have negligible impacts on runoff P losses. To accurately predict changes in P loads or develop strategies to meet P load reduction targets, modeling efforts will need to account for soil P reserves and for how they will change in the future as nutrient application rates are adjusted.

N losses from cropland also have a significant lag-time component where root zone leachate enters shallow aquifers with long residence times. This lag-time needs to be accounted for in modeling efforts to correctly relate the implementation of management practices that reduce root zone nitrate leaching losses to future changes in delivered N loads. In many regions of the Bay watershed, a major fraction of N losses from cropland occurs via nitrate leaching into shallow groundwater. Subsequent discharge of this nitrate in stream baseflow is a dominant component of delivered N loads in many watersheds (Bachman and Phillips 1996, Staver and Brinsfield 1996, Jordan et al. 1997). Failure to consider the lag-time associated with subsurface nitrate transport will result in unrealistic expectations of when reduction in delivered N loads will be achieved even if implementation efforts are successful and effective.

Seasonal variation in groundwater nitrate concentrations could be evaluated through the use of environmental tracers to quantify age and transit time of water and accompanying proportions of N moving laterally or vertically through time, or likely sources of the N pool (Bohlke and Denver 1995, DeWalle et al. 1997, Plummer et al. 2001, Williard et al. 2001, Lindsey et al. 2003, Bohlke et al. 2007, Green et al. 2008, Groffman 2012, Duffy et al. 2012, Liao et al. 2012). Additionally, simulations should examine the effect of soil saturation on nitrate movement to evaluate the importance of unsaturated soil lag-times as influenced by vegetation type, slope, and infiltration variability (C. Welty, pers. commun.). This could include region-wide pedo-transfer function determinations, if a novel remote sensing approach for soil characteristics could be identified. This entire spectrum of approaches could be examined with a multiple-model approach, through coupled groundwater-surface water models and full energy/vegetation modeling. This approach should employ simple (parsimonious) models and represent different hypotheses for describing nitrate transport mechanisms. Ideally, this approach would be

complemented with subsequent monitoring to provide a basis with which to evaluate the models' performance and to identify the most likely mechanism(s) controlling nitrate transport dynamics.

As noted above, soil saturation and depth of the water table are critical features to include in modeling land-to-water loadings. Many urban areas, especially, have channelized stream flows resulting in high bank erosion and deeply incised channels, resulting in drier riparian zones and lower abilities to intercept shallow soluble nitrate as it flows to the streambed (K. Belt in Groffman 2012); streams are disconnected from the floodplain as well, reducing sediment (and P) trapping afforded by tightly coupled floodplains and non-incised streams. Denitrification in connected floodplains is high (Harrison et al. 2011 in Groffman, 2012), constituting a substantial permanent export of N that reduces N accumulation in receiving waters. Because similar floodplain areas in incised urban streams are disconnected and drier, less denitrification is likely. Again, geomorphology/LIDAR data are needed to distinguish between these two types of streams in modeling. These disconnected urban floodplain areas will have lower nitrate levels and lower BMP removal efficiencies, since denitrification typically seen in linked floodplains is substantially curtailed (Groffman 2012).

The models need to be able to simulate the influence not only of present land use but also past land use. For example, it is reasonable to expect that the contributions of N and P from suburban land that was recently converted from forest land will be different from the N and P contributions if the land was previously in agricultural uses. Storage of N and P from these previous uses can be expected to influence N and P export for some decades after the land conversion takes place.

Future modeling refinements ought to consider in-stream processes for modifying nutrients and sediments entering the region's low-order headwaters and larger streams, as well as ponds, lakes, and reservoirs. For example, in other watersheds, in-stream sediment dynamics have been important in assisting land managers in reducing sediment. Additionally, new theoretical methods and subsequent model refinements are needed for processes and time scales that control storage and re-suspension of particles in river corridors, as those processes and storage not only affect downstream transport of the suspended particles that partially govern water clarity and hence habitat, but also affect release of nutrients stored in sediments and those re-mineralized from the benthos. The in-stream release of nutrients that accompanies sediment re-suspension constitutes pulsed delivery and can support subsequent algal production, fueling continued nutrient cycling along streams and rivers (nutrient spiraling, Newbold et al. 1981), oxygen dynamics in slower moving lotic or deeper lentic systems, and eventual loads of nutrients entering tidal waters.

By incorporating these processes into future versions of CBWM or complementary models to inform CBP models (such as the dynamic SPARROW modeling that incorporates N storage in terrestrial vegetation for estimating seasonal export of the nutrient; Smith 2012), it could be feasible to explicitly evaluate alternative management options. In the case of managing stream

sediment loads, the model should incorporate knowledge of fluvial sediment transport to predict what or where management efforts should be targeted to reduce sediment loads delivered to the sub-estuaries. Hence, spatially-explicit models that link local, reach-scale processes across a watershed setting could identify site-specific BMP implementation to maximize water quality benefits.

4. Are there modifications (perhaps post processing) of existing watershed models that could adjust their results to better accommodate lag-times? Would implementing these likely be worthwhile?

The CBWM does not consider lag-times. The intent of the model is to represent the fluxes that would be achieved, at steady state, as a result of the array of activities taking place in the watershed (including the BMPs that have been implemented). The modeled outcomes of the WIPs are intended to represent the steady-state outcomes of the actions that have been taken in the watershed. Requirements under the TMDL are to implement those actions that can be expected to lead to the desired water quality goals. For this reason, the lack of lag-times in the model is not a problem. However, it can be a problem from the standpoint of model calibration. When the model is calibrated to current estimated fluxes from the various watersheds, the calibration runs use the current array of BMPs. This can lead to inaccuracies in the calibration process, because the current water quality can only be expected to be responsive to the legacy of past actions in the watershed. Finding a way to consider past watershed conditions (land uses and BMPs) is an important challenge for the watershed modeling community given that there is no unique lag-time length that can cover an entire watershed.

In addition, the model needs to be able to represent, for the benefit of managers, a realistic time frame for expected improvements due to the BMPs implemented. The actual water-quality "signal" that can be measured at the mouth of the watershed is very much influenced by variations in weather and the resulting wet and dry conditions, as well as by the BMPs whose influence may be gradual over a relatively long period of time. There needs to be a capability to provide to managers a set of simulation outputs, with and without a suite of BMPs, that show a realistic time line of water quality improvements that should result from implementing the suite of BMPs. Understanding the timing and magnitude of the ultimate improvement from the BMPs is important for managers to understand in light of the very substantial natural variability of these watersheds. These issues do not suggest the need for an overhaul of the CBWM, but rather a way to supplement the calibration method and the outputs derived from it.

The current CBP modeling framework represents BMP nutrient or sediment removal immediately upon implementation, without any lag for maturation, etc. In order to continue effective use of the CBWM and enabling jurisdictions to evaluate likely benefits from BMP implementation, the model enhancements suggested above could be undertaken as parallel modeling activities, to inform continued refinement and revision of the CBWM. One of these

models could be a high resolution model for groundwater flows and loadings (Sanford et al. 2012), with groundwater age distribution for a site incorporated into the model to identify age/source of nitrate (new vs. historic N pools, agriculture vs. other sources). Groundwater distributions and ages would be needed from across the region. Another approach for estimating the importance of lag-times in system responses would be to implement two models in a small watershed, first without lag-times and calibrated to observations; then rerun with lag-times, compare output, and calculate uncertainties for the modeled projections. The magnitude of that uncertainty would indicate the relative importance of lag-times in simulating water quality improvements. Still another option would be to use the model(s) to examine pollutant loading responsiveness to large land changes, i.e., ‗hot spots'. How do the models respond to abrupt land change vs. gradual monotonic modifications of the land?

A range of model types, from relatively simple to complex, could be applied to bridge gaps between local and regional data and objectives. For example, simple lumped-parameter models can be useful for exploring potential contaminant responses and rapid sensitivity analysis in various ‗type settings‖ representing different hydrogeologic units (Jurgens et al. 2012). These tools can be applied to water supply wells, as well as to watershed-scale groundwater discharge. Lumped-parameter models typically do not incorporate spatially-distributed properties or land-surface attributes, which must be addressed with more complex models such as flow simulations with particle tracking (e.g., Sanford et al. 2012). These different approaches are well established for steady-state hydrologic conditions, permitting evaluation of contaminant trends at inter-annual and longer time scales, but they must be modified or coupled with other models to evaluate responses at shorter time scales (daily to seasonal) with highly variable recharge and discharge rates. This is important because changes in watershed behavior may be quite different under different flow conditions (e.g., Hirsch 2012). Relations between contaminant concentration and stream flow are not simple, because the dominant pathway of water through a watershed changes at different time scales, and each pathway has its own chemical characteristics. Further complicating the watershed response will be variability in the distribution of reactions such as nitrate reduction, N and P assimilation, and release from biota and sediment in all components of the hydrologic cycle. Fully comprehensive watershed-scale transport and fate models including groundwater, surface water, and biological processes are being developed and should be applied to selected areas of the Bay watershed where data are available to test them. Although these models require extensive data, their use should be enabled by new developments in continuous water quality sensors, as well as from expanded application of environmental tracer analyses of groundwater and surface water age distributions. Model testing should focus on sites experiencing major local land use and management changes on short time scales, especially where previously increasing pollutant loadings are showing reductions. Historic monotonic changes, such as increasing N fertilizer use after the mid-1900s, provide inadvertent experimental data that are useful for this purpose, but more variety in the historical records of loads and practices will help remove ambiguities in evaluating and

calibrating model parameter values. Meanwhile, as increasingly sophisticated models are developed and applied to more diverse sites, sensitivity analyses illustrating potential lag-time effects could be performed with various existing approaches (e.g., lumped parameter, distributed flow simulation), and it might be worth considering whether these approaches or results could be coupled somehow with the existing CBWM.

Insure Continuous Dialog between Monitors & Modelers

A large need is expanded dialog between those who use monitoring data to assess the variations and long-term trends in water quality from headwaters tributaries down to the Bay and modelers in the CBP. Water quality monitoring and assessment personnel best understand the context of data and should have an opportunity to review modeled system responses based on that data, in order to provide feedback to the modelers on possible explanations of differences between modeled output and the field responses. Observations and models are best developed in an iterative fashion, such that model parameters are grounded in observations and observations can be optimized to provide maximum impact on understanding and modeling. Numerous contrasting examples of field conditions can provide modelers with insights into why some models only achieve a limited success in replicating field observations. In urban systems, storm water culverts and pipes are often cracked and leaking, leading to elevated nutrient and sediment inputs to local streams (Groffman 2012). No-till farming practices have been advocated as critical to keeping sediment and particulate-P in place, yet Staver (2012) has observed substantially higher nitrate and dissolved inorganic phosphorus runoff in spring immediately after fertilizer applications than with conventional tillage. These findings illustrate the need to consider the important differences in modeling the impact of land use change on sediment (with long lag-times) and dissolved-P (which has a much shorter lag-time). Legacy sediments were not included in CBP modeling of sediment inputs to the Bay, yet these relic sediments contribute significantly to receiving waters (Walter and Merritt 2008). Reservoir sediment resuspension is now an important issue, exemplified by the 2012 storm-induced Conowingo sediment discharges described in Hirsch (2012). USGS monitoring of groundwater nitrate levels throughout the region, local groundwater flowpath studies of nitrate transport and reaction in representative watersheds, and subsequent regional modeling of groundwater transport to receiving waters (Sanford et al. 2012) have led to an increased focus on assessing immediate vs. longer term BMP impacts on likely water quality improvements. Local studies in contrasting hydrogeologic settings highlight differences and point to critical watershed characteristics that control contaminant pathways, transit times, and natural attenuation sites (e.g., contrasting agricultural nitrate movement through various coastal plain, karst, and crystalline rock terrains) (e.g., Bohlke and Denver 1995, Lindsey et al. 2003, Bohlke et al. 2007). Wetland/marsh nutrient uptake in the Patuxent may be a large reason for the river's recent water quality improvements, identified through nutrient measurements by Boynton and colleagues (2008). Elevated P fluxes from algal bloom-dominated areas in the Potomac (Seitzinger 1991) have informed sediment fluxes/nutrient

biogeochemistry in the CBP's models. Microbenthic algal uptake of N and P in shallow, lighted flanks of the bay and its tributaries (J. Cornwell, pers. commun.) have also modified nutrient recycling and fluxes to the water column in CBP models. These few examples indicate that frequent, continuous dialog between those making and synthesizing field observations and those modeling the bay's responses must typify CBP operations, rather than remaining irregular and as needed. Additional synthesis reports on some of these topics might be warranted, with specific dialog thereafter to identify model adaptations for these critical processes.

5. Are there some broad general statements that STAC can make to the Bay community about the typical lag-times for sediment, nitrogen, and phosphorus associated with broad categories of BMPs that would be applicable over the entire Bay watershed (or significant portions of the Bay watershed)? Is it even useful to try to do this?

Importance of Providing some General Statements about Lag-times

It is important, as well as useful, to make some general statements about typical lag-times, because the general public and policy makers should have realistic expectations for the time-line of detectable water quality improvements that will eventually occur through the intense partner restoration efforts throughout the watershed. This is critical for maintaining public and political support to restore the Bay.

Overview of Lag-times

Lag-times are part of all natural ecosystems. Lag-times vary widely for different types of pollutants. For example, soluble nutrients like nitrate behave differently than sparingly soluble nutrients like phosphorus or organic matter, and suspended sediment behaves differently than nitrogen or phosphorus (Ator et al. 2011, Boesch et al. 2001). Lag-times also vary for different modes of transport. For example, lag-times will be different for storm-driven surface runoff transport compared to base flow through aquifers (Meals et al. 2010, Boesch et al. 2001). Therefore, general statements about lag-times will have to be structured within the context of type of pollutant and mode of transport (as shown in Tables 1 and 2).

Lag-times also have a human or social science component that can be manifested in the need for technical training, or long-term education and cultural change (Meals et al. 2011). Another social component is the development of political will and policy to engage in solving challenging environmental issues. Although these human and social science components were a secondary focus of this workshop, they are, nevertheless, real and need to be recognized.

Estimating General Lag-times within the Bay Watershed

Lag-times are location and scale dependent, i.e., they will vary by physiographic region within the Bay watershed (e.g., Coastal Plain vs. Piedmont Carbonate vs. Ridge and Valley, etc.) and will vary within a watershed with areas near streams having shorter lag-times than areas further from streams (Sanford et al. 2012). Lag-times are also pollutant specific, i.e., they will vary for nitrogen vs. phosphorus vs. sediment within the Bay watershed (Ator et al. 2011, Hirsch 2012). These variations will limit the applicability of general lag-time statements. Therefore, the general statements presented here are not meant to be directly transferable to specific areas or specific pollutants within a sub-watershed, state, or region. Instead, the general statements provide broad general comparisons of lag-times for nitrogen, phosphorus, and sediment.

General Characteristics of Lag-times within the Bay Watershed

Lag-times and restoration effects will be noticed more quickly at the local scale than at the larger watershed scale, due to fewer in-system storages and shorter travel times from source to water body. Conversely, lag-times and restoration effects will be noticed more slowly at the regional and Bay watershed scale, due to many in-system storages (e.g., regional aquifers, dam reservoirs, forests, wetlands, etc.) and longer travel times from source to water body. Thus, one approach to cope with long lag-times in larger watersheds is to emphasize implementation and monitoring on smaller upstream sub-watersheds that are closer to pollutant sources and nearby streams.

Lag-times are different for nitrogen, phosphorus, and sediment. Lag-times for dissolved nutrients (e.g., nitrate) that are primarily transported by leaching through soil and into groundwater will be larger (e.g., several decades), similar to the transport time for water within an aquifer (Phillips and Lindsey 2003, Sanford et al. 2012). Lag-times for sparingly soluble nutrients (e.g., phosphorus, organic matter), which are primarily transported through storm-driven surface runoff (Ator et al. 2011), will be moderate (e.g., several years to a few decades). Lag-times for sediment will be larger than nitrogen or phosphorus (e.g., many decades) because sediment involves suspension of particles of various sizes and storm-driven transport from a source into streams or reservoirs, followed by episodic re-suspension and further transport downstream (Hirsch 2012, Pizzuto 2012). In addition, finer sediment particles and their attached nutrients/pollutants remain in suspension longer than coarser particles, which lead to shorter lag-times within a stream system and faster delivery to downstream reaches for fine particles compared to coarser particles. An important consequence of these varying lag-times for different pollutants is the need for careful evaluation when interpreting water quality data, which is used to calibrate the Bay Model.

Best management practices and improvements in treatment/industrial plants will affect the delivery of nitrogen, phosphorus, and sediment differently due to differences in their respective mode of transport (as illustrated in Table 2). Therefore, the variable scale of lag-times associated

with BMPs and improvements in treatment/physical plants will lead to uncertainties as to when managers, policy makers, or the general public are going to see observable benefits from these restoration practices. One strategy to deal with the BMP lag-time uncertainties is to consider the relative BMP lag-times during site selection, with emphasis given to BMPs with shorter lag-times, if results are desired quickly, and particularly if resources are limited.

General Comparison of BMP Lag-times and Pollutant Effectiveness

Several of the core points from the workshop on ―question five‖ are summarized in Tables 1 and 2. Table 1 compares the general lag-times by pollutant, transport media, and local stream vs. Bay receiving waters. Table 2 compares delivery times between BMP completion and the receiving stream for various BMPs, and rates of the various BMPs for their effectiveness on nutrients and sediment remediation. These general comparisons are not intended to be used for decisions within specific regions or physiographic provinces within the Bay watershed, but instead are intended to compare the relative delivery times for the various BMPs and their relative effectiveness in reducing loads of N vs. P vs. sediment.

Table 1. Summary of relative transport processes and general lag-times by pollutant type, mode of transport, and receiving water body.

Pollutant Type	Transport Processes		Lag-Times	
	Mode of Transport	Relative Importance, scale 1-10 (low-high)	Source to Stream (years)	Stream to Bay (years)
Nitrogen	Ground Water	8	5 - 30	1 – 3
	Surface Water	2	<1 – 5	1 – 5
Phosphorus	Ground Water	1	5 – 30	5 – 50
	Surface Water	9	<1 – 5	5 – 100
Sediment	Surface Water	10	<1 – 5	5 –>100

Table 1 illustrates that nitrogen is primarily transported through groundwater (Meisinger et al. 2008) with the largest lag-time the transport through aquifers to local streams (Sanford et al. 2012). Phosphorus is primarily transported through surface runoff (Ator et al. 2011) which moves phosphorus to the stream within about 5 years, followed by larger lag-times within streams to the Bay. Sediment is only transported by surface water and has the longest lag-time within streams as it moves downstream to the Bay in repeated cycles of suspension and deposition associated with storms (Gellis et al. 2009, Banks et al. 2010, Devereux et al. 2010, Hirsch 2012). Table 1 also illustrates the large relative difference in lag-times between nitrogen, phosphorus, and sediment. For example, the general lag-time range, from source to the Bay, for nitrogen would be a half-decade to a few decades, while the corresponding range for phosphorus

would be half-decade to many decades, with a corresponding range for sediment a few decades to a few centuries.

Table 2. Classification of lag-times for selected management practices or structures and relative effect of BMP for nitrogen, phosphorus, and sediment (see footnote for further details).

Management Practice, Structure, or Upgrade	Lag-Time Class[1] (V. Short-Long) between BMP Installation and Observed Effects In-Stream – and Relative Effect[2] (1-10) on N vs. P vs. Sediment		
	Nitrogen (N)	Phosphorus (P)	Sediment
Non- Point Source, Short Maturation Time (less than 1 year)			
Livestock Exclusion	V. Short - 7	V. Short - 8	Short - 8
Grass Cover Crops	Medium - 8	Medium - 1	Medium - 3
Agriculture Nutrient Management – Basic	Medium - 4	Medium - 4	NA
Agriculture Nutrient Management - Improved	Medium - 6	Medium - 6	NA
Stream Bank Stabilization	V. Short - 2	V. Short - 5	V. Short - 8
Urban Sediment Pond	Short – 3	Short - 5	Short - 8
Non- Point Source, Long Maturation Time(1-10 years)			
Conservation Tillage	Long – 2	Short - 5	V. Short - 8
Grass Buffers	Medium - 2	Short - 5	Short - 8
Riparian Forest	Medium - 4	Medium - 5	Medium – 5
Soil Conservation Plan	Long – 1	Medium - 3	Medium - 8
Urban Nutrient Management	Long – 5	Medium - 5	NA
Point Source, Long Maturation Time (greater than 3 years)			
Treatment Plant N Upgrade	V. Short - 9	NA	NA
Treatment Plant P Upgrade	NA	V. Short - 9	NA

[1]Lag-time class: V. Short <1 yr., Short = 1-3 yrs., e.g., storm flow transport, or near stream location for P and sediment; Medium = 2-10 yrs., moderate distance from stream or shallow ground water; Long = 7-50 yrs., base flow transport of N through aquifer or located in upper half of watershed; NA = not applicable. [2]Relative Effect on N vs. P vs. sediment: scale 1-10 (minor - major effect).

Table 2 relates the general lag-times in Table 1 to some common BMPs that are being deployed for restoring the Bay. Table 2 could be used to identify BMPs that would produce the most rapid response to restoration. For example, practices such as livestock exclusion fencing, stream bank stabilization, urban sediment ponds, and point-source upgrades all have ―Very Short‖ or ―Short‖ delivery times to streams and would be expected to produce in-stream water quality benefits sooner than practices like grass cover crops, riparian forests, or soil conservation plans that require a longer ‗maturation time' between completion and the realization of a water quality benefit at the edge-of-field. In addition, Table 2 provides a general summary of the relative BMP effectiveness for N vs. P vs. sediment. For example, practices like grass cover crops are especially effective for nitrogen mitigation but are less effective for phosphorus and sediment, while stream bank stabilization is least effective for nitrogen (because it is water soluble), but is very effective for sediment control.

Table 2 can also be used for a first-draft comparison of alternative BMPs for nutrient trading. For example, if a treatment plant is considering nutrient trading for upgrading nitrogen removal, it would be beneficial if the landscape BMP being considered for trading had a very short delivery time (similar to the delivery time of the treatment plant upgrade) and had a high effectiveness for nitrogen (such as livestock exclusion), as opposed to a BMP with a longer delivery time (such as cover crops) or a BMP that had little importance for nitrogen remediation (such as grass buffers). Of course this initial nutrient trading comparison would require a more detailed evaluation of the trade, one that would take into account the local hydrology, the baseline BMPs required by the individual state, and other factors.

Tables 1 and 2 are also useful for providing a broad overview of the lag-times expected for the different pollutants and for some common BMPs. One observation from such an overview is that more than half of the lag-times are shown in units of decades or many decades, with lag-times of one-to-three decades common. This clearly indicates that policy makers and the general public need to be made aware of these delays, so that they can adjust their expectations accordingly.

6. Does the consideration of lag-times matter to the implementation of policies such as load allocations or effluent trading? How should these policies deal with the issue of lag-times?

Lags in the delivery of water quality improvements from the adoption of best management practices for improving water quality have several important implications for policies related to the Chesapeake Bay Total Maximum Daily Load.

Trading

One of the most important lag-time considerations is the implication for point/nonpoint trading programs. Point-to-nonpoint trading is expected to play a major role in reducing the overall

costs of meeting the TMDL. Point-to-nonpoint water quality trading allows regulated point sources to purchase reductions from unregulated NPSs. Point sources benefit by meeting the requirements of their discharge permits at a lower cost than if they constructed new facilities, and NPSs benefit by selling an environmental service for a profit. A basic requirement is that the allowance held by the point source and the offset sold by the NPS are ecologically equivalent. This ensures that water quality is not made worse after trading occurs.

―Equivalence‖ is most often discussed in terms of ensuring ecological equivalence, that the environmental impact of the allowance and offset are the same. Trading ratios are mechanisms for ensuring that the NPS offsets purchased by the regulated source have the same environmental impact as the discharge it replaces, with a reasonable degree of certainty (U.S. EPA 2007).

Equivalence also has a temporal dimension. The time frames for buyers and sellers in a trading program must be aligned, in that purchased reductions must be delivered during the same time period that a permit must be met (generally on an annual basis). An implicit assumption in most research on trading is that offsets are delivered in the same year that they are purchased (Ribaudo et al. 2005, Van Houtven et al. 2012). But the evidence strongly indicates that there may be significant lags between when an offset-producing management practice is installed and the delivery of that offset to the point source (National Nonpoint Source Monitoring Program 2008, Meals et al. 2010, Duffy 2012, Sanford 2012, Staver 2012). Such lags have important implications on markets and potentially on water quality.

A simple graph highlights the issue presented by the presence of lagged effects from the source of offsets in a point/nonpoint trading program (Fig. 6). In Figure 6, the supply curve for credits would be Ystatic, if lags in offset delivery are ignored. This curve assumes the edge-of-field reductions are delivered the same year the practices are installed. Equilibrium price and quantity would be Pstatic and Qstatic, respectively. However, because of physical processes, only Q1 is actually delivered in the first year. This means that Qstatic-Q1 offset credits are not actually delivered. Because a portion of regulated sources‗ discharge is not offset, discharge will exceed program goals, the annual permit will not be met, and water quality will be worse than expected. The situation will improve over time as offsets―catch up‖, but in the intervening years delivered offsets will fail to match the allowances held by point sources.

If time lags are accounted for, and permits must still be met annually, the relevant supply curve upon which point sources will base their decision of whether to participate in trading or to install advanced treatment technology will be Yr1. Price of a credit is P1, much higher than Pstatic. Because of the higher credit price, few point sources will opt to purchase credits. Installing enhanced nutrient treatment at a point source facility requires an irreversible capital investment. Once installed, the point source will not enter the market in future years and demand for offsets from agriculture will be permanently reduced.
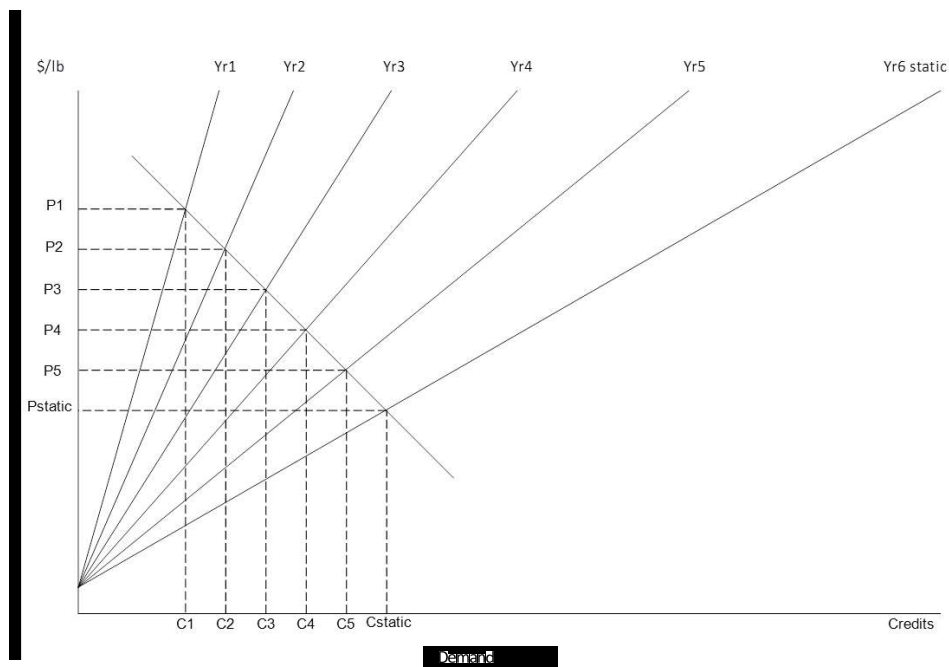
Figure 6. Supply of credits with and without time lags (see text for symbol description).

Well-designed trading programs must include forward markets to efficiently allocate abatement across space and time (Shortle 2012). Since a NPDES discharge permit must be met each year, a treatment plant would like to be able to purchase credits for future delivery. However, forward markets are another source of program complexity. Trading works best when markets are relatively simple with low transaction costs. Point/nonpoint trading is already characterized by complexities of location, stochastic delivery, and uncertainties about BMP effectiveness. The need to also address time lags may mean that plausible markets for achieving the least cost solution cannot be designed because appropriate ecological and economic constraints cannot be made equivalent (Shortle 2012). Markets might still play a role in reducing costs of regulating sources and increasing abatement from NPSs, but expectations on outcomes must be adjusted for the realities of the problem instead of being based only on the—promise‖ of textbook models.

The importance of time lags in trading depends on the extent to which agricultural lands are affected by—significant‖ lags. If only a small proportion of land is characterized by long delivery times, then some simple rules for excluding such land from trading may be an easy course of action. On the other hand, if significant amounts of land are affected, then more careful consideration of lags is warranted. If the amount of agricultural land brought under a trading program is an important program goal (which is implied by EPA guidance for trading),

then allowing short-term degradation of water quality may be seen as an acceptable risk for the benefit of long-term implementation of water quality BMPs on cropland and lower TMDL compliance costs for regulated point sources. However, if any degradation of water quality is unacceptable, then policy approaches other than trading may be warranted.

TMDL load allocation

Another potential issue is how lags might have influenced the TMDL target loads. Water quality monitoring used in the calibration of the CBWM may not be reflective of management practices in place because of lagged delivery of pollutant reductions. From an implementation standpoint, however, this is not seen as a problem. TMDL target loads are set based on load limits from the estuarine model and WIPs are based on the level of BMP implementation, not actual in-stream reduction, even though that is the eventual desired outcome. Both of these critical components of the CBWM are lag-time independent. Where lag-time matters most in the CBWM would have been at the calibration stage, where such information could have been used to improve possible source identification and allocation distributions. But revisiting the model for such a purpose would require data that are probably not available, and could be counterproductive to the progress already made. What can be done is to use information about lag-times to inform the adaptive management process, to educate the public about setting realistic restoration expectations, and to assist local managers in more appropriate selection of control measures that will produce the desired short-term and long-term effects.

7. What, if anything, can or should be done about improving the understanding by the public and public officials regarding lag-times?

The willingness of individual watershed residents to bear the costs of improving water quality depends on the ecosystem and economic benefits that are expected to result. An erosion of the belief that benefits are forthcoming as promised could lead to reduced support for the TMDL-driven programs and policies. The delivery of improved water quality from changes in land/crop management can take years to see because of lag-times. For resource managers, delayed delivery of observable water quality improvements could raise concerns on the part of the public about whether public resources are being—wasted‖, which could lead to a reluctance to continue to support programs for meeting the TMDL, or to support publicly-funded water quality programs in general.

Since lag-time cannot be shortened, the issue becomes how to manage the public's expectations about restoration efforts related to the TMDL. The National Academy of Sciences (NAS) review of program strategies and implementation pointed out that insufficient articulation and explanation of uncertainties in achieving water quality goals due to lag-times and other factors may lead to the public's unwillingness to accept anything but observable improvement (National Research Council 2011). However, the NAS report also presented evidence that adequate

explanation of these lag-times and uncertainties could likely lead to public acceptance, even by those who lack knowledge of technical scientific issues.

Explicit modeling of some sort should be used to create a side-by-side contrasted example that illustrates an expected time-line of nitrogen, phosphorus, and sediment loads delivered to the Bay, with and without consideration of lag-times. Using one or more specific watersheds and specific WIP plans, the role of natural temporal variability and the role of lag-times could be clearly illustrated. The output would show that, after some decades, the two simulations would converge to a single outcome, with the lag effect clearly visible.

Information provided to the public about lag-times could be coupled with measures of progress that are not (as) subject to lags. For example, improved land management could result in more immediate improvements in local water quality, even if improvements are not yet observable downstream. These improvements could be emphasized in progress reports. This may be especially important in parts of the Chesapeake Bay watershed that are bearing a larger share of the restoration costs, but do not border the Bay and would, therefore, not directly benefit from a cleaner Bay.

Public outreach could also include examples of other environmental issues that were subject to regulation followed by long recovery times, such as CFCs, acid deposition, and DDT. Outreach could also include more accessible on-line decision support tools that will enable policy makers to better understand the lags involved and to prepare appropriate education campaigns for the public.

Actions taken to improve water quality might also produce co-benefits, such as improved wildlife habitat or improved recreation opportunities. Such benefits could also be emphasized. It might also be advisable to consider such co-benefits when selecting management options for meeting load allocations, even if they are not the most cost-effective for improving water quality.

Research needs

Research could help all watershed stakeholders to better understand how lag-times might affect the Bay TMDL. Continued research on lag-times from different parts of the watershed would aid in the targeting of management measures, and to better interpret output from the CBWM. This research would also provide insight into the risks that trading programs might pose if lag-times are not included. Research on people's preferences in regards to the delivery of water quality improvements could help in the design of information programs.

Implications and Recommendations

The importance of BMPs in nutrient and sediment reduction is critical to the Bay's restoration, and BMP implementation has become the tracked outcome in the CBP's 2-year milestones.

Hence, identifying BMP maturation and effective operational periods for inclusion in the CBP models should be an expanded priority. Recent CBP model revisions have begun including BMP efficiencies as a function of precipitation amount and age of the BMP and this should continue, and expanding to include landscape type and geographic location (i.e., site specific issues) for use in the model.

One of the most important implications of lag-times is for point-to-nonpoint nutrient trading programs. A basic requirement of trading programs is that the allowance held by the point source and the offset sold by the NPS must be ecologically equivalent both spatially and temporally. If lag-times are accounted for, and permits must still be met annually, the economics driving the decision of point source permit holders about whether to participate in trading or to install advanced treatment technology will favor the advanced treatment technology. Since installing enhanced nutrient treatment facilities requires an irreversible capital investment, once installed, the point source may not enter the market in future years and the demand for offsets from agriculture may be permanently reduced. In order to adjust for the effects of lag-times, existing trading programs could be revised to incorporate forward markets to efficiently allocate reductions over time, but that would add an extra layer of complexity making point-to-nonpoint source trading even more difficult to implement. Alternatively, trades could be restricted to those BMPs known to produce relatively rapid responses in receiving waters.

A major baseline data need for assessing the impacts of lag-time is a comprehensive local inventory of all agricultural and urban BMPs, including performance characteristics. Although this issue has already been highlighted by many Bay Program partners, it bears repeating here, as the effects of any actions related to lag-time are impossible to quantify when the actions themselves have not been fully quantified.

As part of a BMP inventorying process, efforts to enable data sharing from federal sources related to farm-specific BMPs should be expedited, but in a manner that protects client privacy. Nutrient management should also be expanded from a mere planning tool to an accountability mechanism. Measures, previously unaccounted for, such as farm and recreational ponds and stream bank restoration, need to be included as well.

As urban areas begin their BMP inventory efforts for stormwater management, standardization and dissemination of accounting methods, inventory procedures, and performance verification is needed to facilitate widespread and rapid adoption of consistent data to feed into a Bay-wide database.

The current CBWM is a complex model that provides insight into the broad-scale distribution and delivery of nutrients and sediment from a wide variety of sources, but it was never designed to calculate field-scale impacts of site-specific BMPs. Supplemental models should be used to inform the CBWM on processes not currently simulated, to facilitate insights into lag-time, and

to provide site-specific targeting of BMP placement for more effective load reduction. Supplemental models should be used to explore an improved representation of sediment and attached nutrient storages, rather than the current use of sediment delivery ratios to edge-of-stream and the in-stream delivery factors, because sediment storage, rather than transport, appears to be the primary regulator of downstream sediment delivery. This same idea can be applied to nitrogen, phosphorus, and sediment, and implies a need for a mass balance approach for these pollutants. For example, it is the overloading of P in the surface layer of some agricultural soils (particularly those receiving manure that is not worked into the soil) that drives P loss from agricultural lands. This suggests that the direction of research in watershed modeling should take a mass balance perspective rather than just modeling transport with loss terms. Models should consider the sources, the transformations, the long-term and short-term storage, and ultimately transport out of the system. The models also need to consider the transformation of attached particulate nutrients into dissolved forms and their subsequent behavior.

Supplemental models should also be used to explore improved representation of surface-groundwater interchange, as delivery of dissolved nutrients is largely controlled by groundwater flow. Additional topics related to lag-times that should be explored by supplemental models in order to inform the CBWM include the impacts of improved representation of lower order streams, stream channel erosion, and groundwater and reservoir dynamics. Where possible, future revisions to the CBWM should incorporate lessons learned from the supplemental models.

A conceptual framework needs to be developed that encompasses the interactions between floodplains, stream channels, and sediment storages. Major data needs identified in the workshop focused on developing comprehensive spatial and temporal sediment budgets to better quantify the size and distribution of various sediment storages in any given watershed. Tools to help in this effort include particle size distribution studies, isotope tracer studies, sediment coring in reservoirs, use of sediment "fingerprint" techniques to determine the role of various sediment sources, and coupled sediment-nutrient analyses. Research needed at a very local scale includes long-term monitoring at various points along a flow path to the nearest stream (including the impacts of riparian buffers); monitoring of nutrient movement and partitioning between the soil surface, the unsaturated zone, and baseflow in physiographic provinces and landscape settings; and repeat (3-year interval) geo-referenced sampling of phosphorus in the upper few centimeters of soil in fields with high soil test P, as this is the major soil P storage locale, as well as the source of event-driven P delivery. Research needed at the regional scale includes inflow-outflow monitoring of larger reservoirs to improve their representation in the CBWM and groundwater flow path delineation to better understand regional variability in pollutant delivery these sources.

A new focus is also needed in our collective monitoring strategies, consistent with the new conceptual framework that includes the impact of storages on lag-time. There is a distinct difference between results monitored in a demonstration watershed and those expected from a larger geographic area. Funding is scarce for both experimentation and for additional monitoring,

but the general public and legislators demand results and accountability. Given these constraints and demands, we recommend providing guidance to new monitoring efforts to explicitly evaluate hypotheses needed to guide restoration, BMP implementation, and land planning in a holistic manner. Complementary to this strategy, an expanded dialogue is needed between the scientists who assess water quality (through monitoring and data analysis) and modelers, both to improve calibration of the models and to improve the understanding of the changes taking place in the watershed.

Improved communication with the public is needed to distinguish between BMPs that may be expected to show relatively immediate water quality benefits and those whose impact may not be seen for some time. By the same token, additional ecosystem system benefits, beyond those anticipated in the Bay itself, need to be highlighted, such as wildlife habitat and improved recreational opportunities, which also result from implemented BMPs.

Communicating the general magnitude of lag-times for various BMPs is very important for maintaining public support for Bay restoration. We need to emphasize that lag-times are part of all natural systems and that they vary widely for different practices. For example, shorter lag-times are associated with practices that occur close to water resources such as ―livestock exclusion from streams‖ or ―upgrading nutrient removal from treatment plants‖, while longer lag-times are associated with practices that occur farther away from streams and which involve nutrients transported by slower processes (e.g., groundwater and cyclic in-stream deposition/suspension of sediment). Information about lag-times must be used to inform the adaptive management process, to educate the public about setting realistic restoration expectations, and to assist local managers in more appropriate selection of control measures that will produce the desired short-term and long-term effects necessary for Bay restoration.

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Appendix A: Workshop Agenda

Lag-times in the Watershed and Their Influence on Chesapeake Bay Restoration
Scientific and Technical Advisory Committee
October 16-17, 2012
Location: Sheraton Hotel
Annapolis, MD
Meeting Website: http://www.chesapeake.org/stac/workshop.php?activity_id=214

<u>October 16</u>

8:00 am   **Breakfast (Provided)**

8:30 am   **Overview of Workshop Objectives and Agenda - Bob Hirsch (USGS)**
The goal of the workshop is to bring together a diverse set of experts who can suggest ways in which the concept of lag-times can be represented in simulation models of the Chesapeake Bay watershed. The workshop outcome should be a set of recommendations to the Chesapeake Bay Program regarding data collection, research, model development, policy development and public communications that furthers a better incorporation of realistic representations of lag-times in Chesapeake Bay restoration efforts.

8:45 am   **Presentation on how the Chesapeake Bay Program and the various models and strategies in use, currently incorporates lag-times into their process - Gary Shenk (EPA)**

9:00 am   **Lag-times associated with the storage of sediment - Jim Pizzuto (University of Delaware)**
Storage of particles (particularly on floodplains) can create long (decades, centuries, even millennia) lag-times between upland BMPs and response in the Chesapeake Bay. Jim will discuss what we know about lag-time issues and about the scientific approaches to understanding these, leading to improved predictions of the response of a watershed to changes in management practices.

9:45 am   **The use of rare earth elements to quantify fine sediment travel times and distances in small streams - W. Cully Hession (Virginia Tech)**

Cully will describe field experiments being conducted at the Virginia Tech StREAM Lab in Blacksburg, VA to help us better understand the transport, deposition, and re-suspension of fine sediments (<63 μm) during high-flow events. They are injecting sediments labeled with different rare earth elements (REE) during consecutive storms, which allows them to evaluate deposition and re-suspension from flow event to flow event. The REE-labeled sediment has been detected at distances of more than 850 m downstream of the injection site. The knowledge gained from these experiments will improve sediment transport modeling, and help us pick apart the ―lag-time‖ mystery associated with the installation of management practices intended to reduce sediments and associated pollutants to downstream waters.

10:30 am   **Break**

10:45 am   **The use of regional groundwater flow modeling to model transport of nitrate to the Chesapeake Bay and use of the model to explore the timing of potential impacts of changing inputs - Ward Sanford (USGS)**
Ward will describe his use of the USGS MODFLOW model coupled with groundwater and surface water nitrate data to describe the storage and movement of nitrate through the groundwater system of the Delmarva Peninsula. He will also show simulations results displaying the age of water entering the Bay and the flow paths of nitrate to the Bay, and projections of how changes in nitrate input at the land surface would affect nitrate inputs to the Bay and its sub-estuaries over time.

11:30 am   The use of a dynamic spatially referenced regression approach for total nitrogen with consideration of seasonal and long-term watershed storage of nitrogen - Richard Smith (USGS)

Dick will describe a new dynamic approach to SPARROW (Spatially Referenced Regressions on Watershed Attributes) modeling and its application to the Potomac River basin. The model considers the inputs, storage, processes and outputs of nitrogen in the basin and uses statistical parameter estimation. Applications can consider the potential impacts of changes in nitrogen applications and/or changes in climate forcing on nitrogen outputs by sub-watershed over time.

12:15 pm   Lunch (Provided)

1:00 pm   Riparian, stream and floodplain restoration in urban watersheds: the experience from the Baltimore Ecosystem study - Peter Groffman (Cary Institute of Ecosystem Studies)

Peter will describe the results of research on nitrogen processing in streams, riparian zones and floodplain wetlands in urban, suburban and exurban watersheds in the Baltimore metropolitan area. This processing is controlled by the nature and extent of hydrologic and geomorphic alteration associated with urbanization and is quite responsive to restoration efforts that influence hydrologic connectivity, residence time and soil/sediment conditions. A few examples will be shown and issues related to lag-times between completion of the restoration project and the realization of stream-quality improvements will be discussed.

1:45 pm   Temporal dynamics of changes in delivered nutrient loads resulting from various cropland nutrient reduction practices - Ken Staver (University of Maryland)

Ken will present results from long-term field studies on the movement of nutrients from Coastal Plain cropland through both surface and subsurface flow paths after implementation of most of the major nutrient reduction practices.

2:30 pm   Break

2:45 pm   The isotopic age and transit time of natural flow systems - Chris Duffy (Penn State University)

 New approaches to watershed modeling developed at the Susquehanna/Shale Hills Critical Zone Observatory (CZO) are applied to the problem of predicting the dynamic age of stable isotopes, nutrients and other geochemical species in an integrated ground-water surface-water solute transport model. The model is used to evaluate multi-year watershed responses to climatic variations and estimate the relative age of solutes in each hydrologic state.

3:30 pm   Organization and objectives of breakout groups - Jack Meisinger (USDA-ARS)

Participants will be assigned to one of the three groups. The groups will discuss what is well understood about aspects of lag-time issues and what new knowledge and methods are needed.

Each breakout should address each of the following issues:

1. What new data collection, data analysis, and research is needed at the scale of individual management actions (a single BMP implementation)?

2. What new data collection, data analysis, and research is needed at the scale of river reaches, reservoirs, floodplains, wetlands, and aquifers?

3. What new approaches to modeling should be developed and/or enhanced to better understand and predict lag-times?

4. Are there modifications (perhaps post processing) of existing watershed models that could adjust their results to better accommodate lag-times? Would implementing these likely be worthwhile?

5. Are there some broad general statements that STAC can make to the Bay community about the typical lag-times for sediment, nitrogen and phosphorus associated with broad categories of BMPs that would be applicable over the entire Bay watershed (or significant portions of the Bay watershed)? Is it even useful to try to do this?

6. Does the consideration of lag-times matter to the implementation of policies such as load allocations or effluent trading? How should these policies deal with the issue of lag-times?

7. What, if anything, can or should be done about improving the understanding by the public and public officials regarding lag-times?

The three breakout groups would be organized by these 3 areas of interest:
- **A. Processes associated with the erosion, storage and re-entrainment of sediment and associated nutrients (N and/or P).**
  - **Moderators:** Bob Hirsch and Gene Yagow
  - **Location:** Severn Room

- **B. Processes associated with transport, reaction, and storage of N and/or P in their dissolved form in soils, vegetation, shallow groundwater, and across the groundwater/surface water interface.**
  - **Moderators:** Claire Welty and Weixing Zhu
  - **Location:** Glebe Room

- **C. Dealing with lag-times in the context of regulation, enforcement, pollutant trading, and public perception.**
  - **Moderators:** Marc Ribaudo and Jack Meisinger
  - **Location:** Chester Room

4:00 pm   **First meeting of breakout groups (more to follow in morning)**

5:00 pm   **Adjourn for the day**

October 17

8:00 am   **Breakfast (Provided)**

8:30 am   **Consideration of how pollution trading systems can be designed so as to consider the differences in time lags among multiple pollution sources - Jim Shortle (Penn State University)**
Jim will discuss what lags mean for the allocation of pollution abatement across types, time and space. This includes which sources cannot be used to achieve near term goals and the implications for abatement costs. In addition, Jim will discuss what lags mean for the design of trading markets and the potential role for lagged sources.

9:15 am   **Feedback from Day One - Jack Meisinger (USDA-ARS)**
Open discussion of the charge to the breakout groups. Opportunity to seek clarification of information about current Chesapeake Bay Watershed models and regulatory approaches. Organization and objectives of morning discussion groups.

9:45 am	**Breakout groups continue**
10:30 am	**Break**
10:45 am	**Breakout groups continue**
12:00 pm	**Lunch (Provided)**
12:45 pm	**Breakout groups report back (15 minutes each)**
1:30 pm	**Overview of breakout ideas and initial ideas for STAC workshop report and possible review article**
2:30 pm	**Meeting adjourns. Organizing committee stays on to discuss next steps**
3:30 pm	**Organizing committee adjourns**

Appendix B: Workshop Attendees

Presenters

Chris Duffy
Peter Groffman
Cully Hession
Jim Pizzuto
Ward Sanford
Gary Shenk
Jim Shortle
Dick Smith
Ken Staver

Workshop Attendees

Tobias Ackerman
Rich Alexander
Scott Ator
Larry Band
JK Bohlke
Kathy Boomer
John Brakebill
Darrell Brown
Jeff Cornwell
David Costello
Lee Currey
Bill Dennison
Judy Denver
Jack Frye
Allen Gillis
Pierre Glynn
Chris Graham
Ciaran Harman
Matthew Johnston
Tom Jordan
Neely Law

Workshop Attendees (cont.)

Louise Lawrence
Lewis Linker
Beth McGee
Tom Nolan
Scott Phillips
Karen Prestegaard
John Rhoderick
Dave Sample
Katie Skalak
Bill Stack
Kurt Stephenson
George Van Houtven
Guido Yactayo
Qian Zhang

Steering Committee Members

Russ Brinsfield
Matt Ellis
Natalie Gardner
Bob Hirsch
Jack Meisinger
Marc Ribaudo
David Sample
Kevin Sellner
Don Weller
Claire Welty
Gene Yagow
Weixing Zhu

Appendix C: Workshop Presenter Summaries

Presentation on how the Chesapeake Bay Program and the various models and strategies in use, currently incorporates lag-time in their proves – Gary Shenk (EPA-CBPO)

Currently, lag-times are not incorporated into the CBP decision process or the models that guide them. The explicit goal of the Chesapeake TMDL and the WIPs is to have practices in place by 2025 that will eventually lead to attainment of the water quality standards once lags have played themselves out. Generally speaking, the CBP watershed model is not set up to handle lags and care is taken to remove lags that do exist from the scenario process. This follows directly from the management question implied by the TMDL and WIP goal: What is the effect of a particular set of management actions on the long term loads of sediment and nutrients to the tidal Bay.

A short description of the CBP watershed model structure was provided. The process-based model of a representative acre does not incorporate a significant groundwater lag when loading a down-gradient stream model. Low order stream networks are not explicitly simulated. The use of the watershed model in the TMDL was also discussed. For a more detailed description, see the phase 5 documentation
http://ches.communitymodeling.org/models/CBPhase5/index.php
or sections five and six of the TMDL documentation.
http://www.epa.gov/reg3wapd/tmdl/ChesapeakeBay/tmdlexec.html

Although lag-times will likely not be considered in the CBP watershed model's use in evaluating long-term plans, the inclusion of lag-times in the modeling framework could have at least three advantages.
1. Calibration - The watershed model is calibrated to observed stream data which are the product of lag-times so the inclusion of lag-times would theoretically improve the calibration
2. Validation of predictions - The watershed model is used to make predictions about the effect of management actions. When comparing to flow-normalized monitoring data it would be useful to understand the lag-times to help interpret the comparison.
3. Communication – an understanding of lag-times would benefit the messaging to the public about time frames for water quality improvement.

Lag-times Associated With The Storage of Sediment: Progress Towards Developing Conceptual and Quantitative Models – Jim Pizzuto (University of Delaware)

Watershed BMPs are typically used to reduce nonpoint loading of suspended particles (sediment, phosphorus) to receiving watersheds such as the Chesapeake Bay. However, because suspended material can be stored in floodplains and other depositional reservoirs of alluvial valleys, there may be a significant lag-time between implementing a BMP and a resulting positive impact to an estuarine depositional basin. Current geomorphic measurement programs often result in sediment budgets, which successfully document the importance of storage, but provide no information on timescales required to route suspended particles downstream.

Available models of fluvial transport and deposition either neglect storage or are too complex to be used at the scale of large watersheds. Analysis of sediment budget studies in the Chesapeake Bay Watershed, if properly interpreted, can provide useful information on the expected magnitude of lag-times associated with ―tagged‖ suspended sediment particles. For example, along the South River in Virginia, a 4th-5th order mixed bedrock alluvial channel, 2.2 +/- 0.1 % of the annual suspended sediment load is exchanged per kilometer between the channel and alluvial storage reservoirs (the latter include the floodplain, hyporheic zone, lateral migration deposits such as point bars, and fine-grained channel margin deposits that form in the lee of obstructions along the stream's banks). The inverse of this exchange rate provides an estimate of the transport length scale for complete replacement of the suspended sediment load with new sediment from storage reservoirs. For the South River (ignoring tributaries and potential changes in the geomorphic setting with distance downstream), this distance is 1/(0.022 +/- 0.1) or 46 +/- 22 km. This indicates that reductions in sediment loading (or reduced concentrations of contaminated sediments) created by a BMP cannot influence downstream reaches farther than 46 km. Furthermore, once particles enter storage, on average they remain in place for a very long time; for example, an estimate of the storage timescale for the South River is 4800 +/- 2600 years. By combining the transport length scale of 46 +/- 22 km and the storage timescale of 4800 +/- 2600 years, a time averaged velocity of downstream suspended sediment movement that includes time spent in storage can be obtained, and the resulting value is 9 +- 7 m/yr. At this velocity, sediment related―effects‖ will take thousands of years to travel from the uplands of the Chesapeake Bay Watershed to the Bay itself. This estimate, of course, is for an―average‖ suspended sediment particle, and it also assumes that the geomorphic setting along the stream is uniform, whereas mid-Atlantic channels are typically characterized by alternating transport and storage reaches. Additional research is needed 1) to quantify grain size effects (because smaller particles should travel much farther and faster than larger particles), 2) to document the spatial extent of storage and transport reaches in the Chesapeake Bay Watershed, and 3) to develop improved modeling tools that can adequately represent these processes.

Quantifying Sediment Transport and Fate in Small Streams Using Rare Earth Elements as Tracers ‒ W. Cully Hession (Virginia Tech)

Presentation by: W. Cully Hession; Collaborators: Tyler Kreider (VT-BSE), Kevin McGuire (VT-FREC), Tony Buda (USDA-ARS), Danny Welsch (CVI).

If we stop a sediment particle from falling in the stream along Craig Creek near Blacksburg, VA through implementation of some management practice, when might we expect to see an improvement in the Chesapeake Bay due to this reduction in sediment?

Or, in other words, how long does it take a sediment particle to make its way down ~135 km of Craig Creek to the James River, where it would need to travel another ~ 560 km to the Chesapeake Bay?

We really don't know enough about sediment fate and transport to answer these questions. The problem is actually even more complex than when we consider thousands of sediment particles that might enter the stream (or not) and would all travel separate paths to the Bay (or not). Our overall goal is to quantify in-stream sediment travel time (or lag-time) by answering the following questions using field experiments and modeling: 1) Where does it go? 2) How far does it go? 3) How long does it take to get there? and 4) If we stop the sediment from getting into a stream, when will we see improvement downstream?

We determined that rare earth element (REE)-labeled sediment was a viable tracer at the reach-scale within a 2nd-order stream at the StREAM Lab (http://www.bse.vt.edu/site/streamlab/) in Blacksburg, VA. Streambank soil was labeled with a unique REEs for injection during two storm events. Suspended and bed sediment sampling occurred during and after each storm event at intervals along the 875 m reach. Over 38% of the injected REE was accounted for in the suspended sediment after 250 m. The tracer was also detected in the suspended and bed sediment samples at the maximum sampling distance of 875 m. Subsequent storm sampling showed no resuspension of previously injected tracers, indicating that the tracer either remained suspended during the first storm or is permanently deposited within the reach. REE-labeled sediment holds great promise for linking soil erosion and sediment fate and transport in both terrestrial and fluvial settings. In the future we will focus on additional injections with expanded longitudinal sampling to better quantify how far the labeled sediment travels and how much of it gets there. In addition, we plan to utilize REEs to label a stream bank or other near-stream areas *in-situ* so that the link between terrestrial erosion and fluvial transport can be better understood.
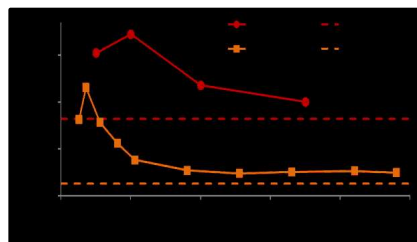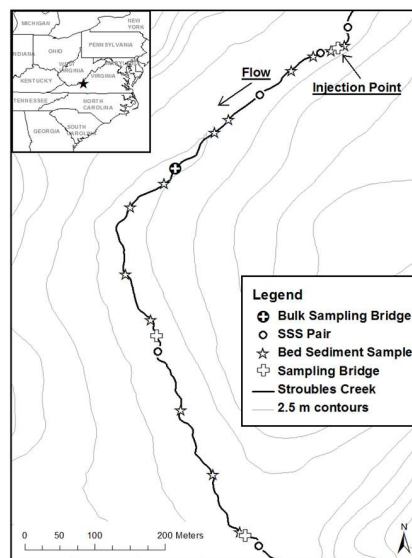
Using a New Groundwater-Regression Model to Forecast Nitrogen Loading from the Delmarva Peninsula to the Chesapeake Bay – Ward Sanford (USGS)

Nitrogen is a major pollutant to the Chesapeake Bay, and EPA is currently establishing total maximum daily loads (TMDLs) across the bay watershed to try to limit or reduce loads of nitrogen to the bay. Many of the current watershed models (for example HSPF and SPARROW) can reproduce current loads to the Bay and estimate their spatial distributions, but they cannot take into account the large amount of dissolved nitrogen in storage in groundwater and the distributions of groundwater travel times. Therefore they cannot predict the temporal response of nitrogen outflows from watersheds to changes in nitrogen loading at the land surface.

A groundwater model was developed in this study of the Eastern Shore (the Maryland and Delaware drainages to the Bay on the Delmarva Peninsula) that calculates distributions of groundwater travel times across the region. These travel times were coupled with a stream-nitrate, mass-balance regression model and fit to spatial and temporal data from seven different watersheds within the Eastern Shore. The data indicate smaller increases in stream nitrate over the last few decades, and the model suggests these smaller increases could be the result of the implementation of best management practices (BMPs).

The parameterized regression model was applied to HUC-11 watersheds across the region to forecast changes in the total nitrogen flux from the Eastern Shore to the Bay. These fluxes include estimates from both base flow and high flow conditions. The EPA has established a target for reduction in the TMDL from the Eastern Shore of approximately 3 million pounds of nitrogen per year. Results from this new model suggest that this target is unlikely to be reached for decades due to the very sluggish response time of the groundwater system. The calibrated regression equation has also been used to create maps that can help target the most effective areas for future reductions in loading at the land surface.

Use of Dynamic SPARROW Modeling in Characterizing Time-Lags in Nitrogen Transport in the Potomac River Basin – Richard Smith (USGS)

SPARROW models are widely used to identify and quantify the sources of contaminants in watersheds and to predict their flux and concentration at specified locations downstream. Conventional SPARROW models are statistically calibrated and describe the average (―steady-state‖) relationship between sources and stream conditions based on long-term water quality monitoring data and spatially-referenced explanatory information. But many watershed management issues stem from intra- and inter-annual changes in contaminant sources, hydrologic forcing, or other environmental conditions which cause a temporary lag between watershed inputs and stream water quality. This presentation described a dynamically calibrated SPARROW model of total nitrogen flux in the Potomac River Basin using seasonal water quality and watershed attribute data for 80 monitoring stations over the period 2002 to 2008 (28 seasonal time steps). The spatial reference frame of the model is a 16,000-reach, 1:100,000-scale channel

and catchment network. The SPARROW model includes five contemporaneous source terms: point source discharges, fertilizer application, manure production, urban runoff, and atmospheric deposition. A sixth source term, representing the delayed release of nitrogen from previously-stored nonpoint inputs, is based on an estimate of the seasonal lag-1 concentration of total nitrogen. Model terms affecting nitrogen transport to stream channels include runoff, seasonal change in runoff, and Enhanced Vegetation Index (EVI) data from the Terra Satellite-borne MODIS sensor used to parameterize seasonal uptake and release of nitrogen. Calibration of the model was conducted with nonlinear regression. All but one of ten estimated coefficients (5 N sources, 3 land-to-water factors, 1 storage term) were highly statistically significant ($p<0.0001$). The coefficient for seasonal EVI was especially strong (t=9). The rate of stream decay was estimated to be small (approx. 0.01 per day) and the coefficient was not statistically significant. Overall R^2 was 0.90. The average source share for the storage term was 51%, suggesting that about half of the yield in an average watershed comes from nitrogen stored for longer than one season. For many watersheds, the stored fraction was greater than 70%. Estimated residence time for catchment nitrogen varies from a fraction of a year to several years depending on location and hydrologic conditions. The model can be used to explore the short- and long-term response of total nitrogen flux to changing precipitation under either constant or changing nitrogen inputs.

This presentation also described calibration of a spatially homogeneous model of the Upper Potomac Basin using longer-term (1973-2010) data on seasonal basin inputs and outputs. ―Observed‖ nitrogen flux estimates at Chain Bridge were regressed on estimated total sources, temperature, and streamflow. Calibration was highly successful. In this model, estimated residence time varies from a fraction of a year to more than 20 years depending on streamflow and temperature.

Riparian, stream and floodplain restoration in urban watersheds: the experience from the Baltimore Ecosystem study ‒ Peter Groffman (Cary Institute of Ecosystem Studies)

With contributions from the scientists and support staff of the Baltimore Ecosystem Study urban Long Term Ecological Research project (NSF DEB 1027188).

In the Baltimore Ecosystem Study, one of two urban long-term ecological research (LTER) projects funded by the U.S. National Science Foundation, we are using―the watershed approach‖ to integrate ecological, physical and social sciences (Pickett et al. 2011). The project produces useful data on nitrogen and phosphorus loads from urban and suburban watersheds to the Chesapeake Bay (Groffman et al. 2004, Kaushal et al. 2008a, Shields et al. 2008, Kaushal et al. 2011, Duan et al. 2012) and detailed information on processes in urban streams, riparian zones and home lawns.

Watershed input/output budgets for nitrogen (N) have shown surprisingly high retention which has led to detailed analysis of sources and sinks in these watersheds. Home lawns, thought to be major sources of N in suburban watersheds, have more complex coupled carbon and N dynamics than previously thought, and are likely the site of much N retention (Raciti et al. 2008, Groffman et al. 2009, Raciti et al. 2011a, Raciti et al. 2011b, Raciti et al. 2011c). Riparian zones, thought to be an important sink for N in many watersheds, have turned out be N sources

in urban watersheds due to hydrologic changes that disconnect streams from their surrounding landscape (Groffman et al. 2002, Groffman et al. 2003, Gift et al. 2010). Geomorphic stream restoration designed to reverse structural degradation caused by urban runoff can increase in-stream retention by creating instream features with high denitrification potential (Groffman et al. 2005, Klocker et al. 2009, Harrison et al. 2012a), by reconnecting streams and riparian zones (Kaushal et al. 2008b) and/or by creating floodplain wetlands (Harrison et al. 2011, Harrison et al. 2012b). Stormwater detention basins may function as ―hotspots‖ of denitrification, restoring functions lost to urban riparian and stream degradation (Groffman and Crawford 2003, Bettez and Groffman 2012).

Considering the ―human element‖ is critical to improving the environmental performance of urban and suburban ecosystems. Understanding why and how people manage their lawns is critical to reducing the water quality impacts of this dominant suburban cover type (Zhou et al. 2009). Including human goals in stream restoration can improve support for restoration projects and encourage become residents to become monitors and advocates for stream ecosystem integrity (Groffman et al. 2003). Watershed restoration can catalyze socio-economic revitalization in economically troubled urban neighborhoods (Hager et al. 2012).

Temporal Dynamics of Changes in Delivered Nutrient Loads Resulting from Various Cropland Nutrient Reduction Practices – Kenneth Staver (UMD)

Cropland is a major land use in the Chesapeake Bay watershed and has been identified as a primary contributor to excessive nutrient inputs to the Bay. Efforts have been underway for more than two decades to reduce sediment, N and P losses from cropland. How many of the most widely implemented practices affect edge-of-field sediment and nutrient losses has been studied for nearly three decades in field-scale watersheds at the UMD-Wye Research and Education Center located in the Maryland Coastal Plain. While there are distinct seasonal patterns of discharge driven by seasonal patterns of evaporation and rainfall intensity, there also is a high degree of short-term variability discharge from cropland due to deviation of rainfall patterns from long-term average values. Thus, management practices reduce the potential for losses, but actual losses in the short-term are driven by the timing of excess precipitation relative to the availability of soluble forms of N and P and erodible sediment.

Reducing the intensity of tillage reduces the potential for field sediment losses immediately with the greatest impact being in the immediate post tillage period, which is usually in the spring prior to planting of summer annual crops (e.g., corn and soybeans) and early autumn prior to planting of winter annual cereals or hay crops. The reduction in the potential for edge-of-field sediment and sediment-bound nutrient losses will occur as soon as reduction in tillage intensity occurs, with the extent of the reduction dependent on the erosion potential of the site, and the intensity of precipitation in the post tillage period. At sites such as the Wye watersheds, which drain almost directly into tidal waters, the reduction in edge-of-field sediment losses translates almost directly into equivalent reductions in loads delivered to tidal waters. In areas of the Bay watershed more distant from tidal waters, the temporal relationship between edge-of-field reductions in sediment loads and delivered loads will be less direct due to the

complex processes of deposition and erosion of sediments in stream systems (see Pizzuto presentation).

Like sediment losses, the potential for surface runoff losses of N and P can be altered rapidly with management practices. Runoff dissolved nutrient concentrations are largely controlled by the availability of water-soluble forms of N and P on or near the soil surface. Most inorganic fertilizers contain highly soluble forms of N and P and most organic nutrient sources contain significant fractions of soluble nutrients. If left on the soil surface, applied nutrients create the potential for extreme spikes in surface runoff nutrient concentrations that can increase edge-of-field nutrient losses early in the summer growing season. The impact of injecting nutrients or using tillage to incorporate them into the soil immediately reduces the potential for extreme spikes in surface runoff dissolved nutrient concentrations. Even more so than the case for sediment, edge-of-field surface runoff N and P losses reach receiving waters quickly during runoff events so any practice that reduces N and P concentrations in runoff will have nearly an immediate effect on delivered loads to tidal waters, depending on the location of field. While the method of nutrient application has the potential to rapidly alter surface runoff N and P concentrations, in the long-term, edge-of-field runoff P concentrations are highly influenced by soil P levels. The concentration of livestock and poultry production in many regions of the Chesapeake Bay watershed, and the resulting generation of large quantities of P-rich manures have led to the buildup of cropland soil P reservoirs in concentrated animal production areas. This build up occurred over a period of many decades and in some areas the increase in soil P is equivalent to decades of crop P removal. Where soil P levels have reached very high levels, they will be the dominant factor controlling runoff P concentrations on an annual basis even if erosion is effectively controlled. In these settings, reducing runoff P losses in the long-term will require drawing down soil P concentrations through crop removal. This process will take many years, and in more extreme cases many decades, depending on the magnitude of the historical buildup of soil P. A rough approximation is that crop removal by most grain crops can only be expected to reduce soil P Fertility Index Values (FIV) about 5-10 units per year in the complete absence of P applications. Since the distribution of soil P concentrations is not well defined spatially in most watersheds, nor are application methods systematically tracked, it is hard to know the fraction of runoff P losses driven by short term management actions versus soil P concentrations. This balance between short and long-term drivers of P losses will need to be characterized before the trajectory of reductions in P losses can be accurately predicted under various management strategies.

There also is potential for long delays between implementation of management practices that reduce root zone N availability (e.g., nutrient management and cover crops) and when delivered N loads respond but the reasons for the lag are different than those for P. Nitrate-N is the primary soluble form of N available for leaching and several practices are available for reducing nitrate concentration in relatively short time frames. However, in many settings, the primary route of N losses from crop land is via leaching of nitrate into shallow groundwater. Root zone leachate can take several years to reach the underlying aquifer where it can reside for years and even decades before discharging into surface waters depending on aquifer characteristics and location relative to discharge zones. (see Sanford presentation). This lag-time can play a major role in overall N loading rates since nitrate leaching often represents a major

fraction of total N losses from crop land. For example in the Wye study watersheds, leaching of nitrate-N was 5-10 times greater than total N losses in surface runoff before the use of cover crops was initiated. Groundwater nitrate discharge is generally the dominant fraction of watershed total N loads throughout the Coastal Plain. Because of the long groundwater residence times in many watersheds, stream N loads are not directly related to current management activities. Where aquifer storage capacity is large, the near term (< ten years) trajectory of stream N loads primarily will be a function of existing aquifer nitrate levels resulting from past land use practices. Nevertheless, where discharge of nitrate enriched groundwater is a dominant component of total N loads, reducing N loads will require ongoing implementation of practices that reduce nitrate leaching and realistic expectations for when watershed N loads will change.

In summary, overland flow transport of sediment, N and P can be altered rapidly by tillage practices and how soluble nutrients are applied. Soil P runoff losses will be less amenable to short term management changes where soil P reservoirs have been built up to very high levels due to long-term P applications in excess of crop removal rates. In these settings, decades may be required to gradually draw down soil P levels and resulting P losses. Delivered N loads also can take decades to change where nitrate leaching is the dominant N loss pathway and where shallow aquifers contain large volumes of groundwater relative to annual recharge/discharge volumes.

The Isotopic Age & Transit Time of Natural Flow Systems – Christopher Duffy (PSU)

The concept of ―age‖ in terrestrial watersheds and river basins has long been a useful quantity for the analysis of process time scales (Phillips 1995) and resource assessments (Allison and Hughes 1973), and recent reviews of the modeling and experimental strategies has greatly organized our approach to the problem of age of waters (Kazemi et al 2006, IHP-V 2001, Brooks et al 2010, Bhatt 2012). In this presentation a model for the age of solutes in watershed flow systems governed by transient flow dynamics is presented along with experimental data currently being compiled for application of the model in the case stable isotopes of water, d2H and d18O at the Shale Hills Critical Zone Observatory. The spatiotemporal distribution of isotopes across landscapes or isoscapes represents a new way of ―seeing‖ our environment that supports the predictive understanding of the patterns and processes that determine isotopic age in complex environmental systems. In this paper we extend the notion of isotopic age for water to include the biogeochemical cycles of the catchment systems. We note extensive data already exists to test the model and the concept outlined here provides a basis for predicting flow paths, residence times and the relative age of water for fluid parcels and biogeochemical processes within the catchment system.

Economics and Nutrient Reduction Instruments: Implications of Lags – Jim Shortle (PSU)

Economic efficiency is a goal of policies for reducing nutrient discharges to water resources. When polluters‘ emissions have equivalent impacts on ambient water quality efficiency is achieved when abatement is allocated across dischargers to equalize the marginal costs of control. With perfect information on abatement costs, a regulator could efficiently

allocate pollution control across polluters. However, regulators do not have perfect information. The advantage of a trading program is that a regulator does not need polluters' private cost information. An efficient solution can be achieved by setting a discharge cap for a watershed, allocating discharge allowances equal to the cap to dischargers, and allowing polluters to trade allowances. With enough traders and perfect competition the cap will be met at least cost.

The textbook model assumes emissions have equivalent impacts regardless of the location of the discharge. In reality, location matters. A smaller fraction of upstream discharge will reach the Bay than downstream discharge, so abatement at different places is not equivalent. Such locational differences can be accounted for with trading ratios, but this makes trading, more complex to implement.

Time creates another factor complicates policy design. One way that time enters is in capital investment decisions. A sewage treatment plant must decide whether to invest in a new treatment technology now or to purchase credits in the market now and in the future. A farmer must decide whether to invest in BMPs to sell offsets in the future or to sell the farm. Their answer will depend on expected future prices. Well-designed trading programs must include forward markets to efficiently allocate abatement across space and time. Since a discharge permit from EPA must be met each year, a treatment plant would like to be able to purchase credits for future delivery. Forward markets are another source of program complexity.

Time lags in pollutant delivery lead to further complications. Evidence suggests that the adoption of a BMP may not produce reductions in discharges in the tidal waters of the bay for years or even decades, depending on the location in the watershed, soils, and the pollutant. The efficiency rule now requires that marginal cost of abatement is equalized across sources with equal time lags, and that marginal cost of abatement of lagged sources in any one time period is equal to the discounted present value of the marginal cost of abatement of nonlagged sources in the year in which lagged pollutant reaches the bay. The practical implication for a policy is that in the short term pollution abatement from unlagged sources will be favored, as the quantity of abatement from lagged sources will be limited, and the price high. Over time, as the number of credits from lagged sources—catch up‖, prices will fall and lagged sources will play a greater role in overall abatement. However, where capital adjustment costs are high, the lack of delivered abatement in early years may trigger the adoption of treatment technology rather than the purchase of offsets, permanently reducing the demand for credits delivered in the future.

Nonpoint source pollution adds even more complexity to the market, as it is stochastic due to weather and BMP performance is uncertain. Taken together, the complexities of location, time, and NPS uncertainty may mean that plausible markets for achieving the least cost solution cannot be designed because appropriate ecological and economic constraints cannot be made equivalent. Markets can still play a role of reducing costs of regulating sources and increasing abatement from nonpoint sources, but reliance on markets and expectations must be adjusted for the realities of the problem instead of the—promise‖ of textbook models.

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